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16. ABSTRACT

Two questions were asked: (1) Will septic tank diversion reduce symptoms of eutrophication in the Twin Lakes, two similar dimictic lakes, and (2) Will a maximum hypolimnetic dose (residual dissolved aluminum $\leq$ 0.05 mg/l) of aluminum sulfate (alum) reduce sediment phosphorus (P) release and further improve them?

After diversion, P in lakes and streams declined and plankton biomass declined, but during 5 years of monitoring after diversion, P and biomass remained those of moderately eutrophic lakes, and P income remained above "permissable". Non-point sources were important.

In laboratory and field tests, maximum dose, which is related to alkalinity and determined by "jar tests", was effective in retarding P release from sediments. A pilot treatment of Dollar Lake in 1974 was done as a test of the technique. 139.55 M^3 of alum were added to the hypolimnion of West Twin (WTL) Lake in July, 1975. Compared to a control lake, hypolimnetic P was lowered in 1976 but net internal P loading, while reduced, continued, suggesting littoral sources. Epilimnetic P also declined in WTL, but remained within the eutrophic range.

WTL improved, as indicated by a trophic state index, but remains moderately eutrophic. No effects on fish were observed.

Details for dose determination and application are given.

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EFFECTS OF DIVERSION AND ALUM APPLICATION ON TWO EUTROPHIC LAKES

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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL), in Oregon.

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report provides a valuable contribution to knowledge of lake restoration and eutrophication control especially in areas of diversion and in-lake treatment.

A.F. Bartsch
Director
Corvallis Environmental
Research Laboratory

ABSTRACT

Two questions were asked: (1) Will diversion of septic tank drainage from East (ETL) and West (WTL) Twin Lakes, two shallow (<5M), morphometrically similar, dimictic lakes in Northeastern Ohio, bring about reduced symptoms of eutrophication and (2) Will a maximum hypolimnetic dose of aluminum sulfate to WTL, defined as the amount of alum which can be added before residual dissolved aluminum exceeds 0.05 mg Al/l, reduce internal phosphorus (P) loading and further improve the lake?

A return to normal fecal coliform levels in the study streams and lakes occurred after diversion. While P in streams was reduced, income did not fall to an "acceptable" level (Vollenweider, 1976). Groundwater was found to be an important P source. P in the lakes was lowered, and symptoms of eutrophication, measured as algal standing crop, were reduced just after diversion (1972), but P and biomass levels remained within ranges expected of moderately eutrophic lakes. Had diversion not occurred, the lakes would probably be closed to recreation due to excessive coliform levels and be classified as very eutrophic.

The P content of the lakes varied each year from a late winter high to a pre-thermal stratification low, followed by an increase over the summer, up to 100% of which was due to net internal P loading. Based on the hypothesis that this loading was hypolimnetic in origin, 139.55 M^3 (36,919 gals) of alum were added to the hypolimnion of WTL in July 1975, leaving ETL as a downstream control. Alum was earlier shown to be effective in retarding anaerobic sediment P release for up to two years in Dollar Lake, and up to 104 days in _in situ_ columns. The dose as defined was shown to be non-toxic to fish and to certain littoral invertebrates.

Hypolimnetic P concentration in 1976 was greatly lowered, but net internal P loading, while reduced, continued at a significant rate, strongly suggesting the littoral as an internal P source.

Epilimnetic P was also lowered, and a Trophic State Index indicated some reduction in symptoms, but WTL remained eutrophic in 1976. No adverse effects or excessive concentrations of aluminum, sulfate, or hydrogen ion were observed.

The treatment may continue to be effective, but could be overcome by continued excessive loading resulting from poor land management.

It is suggested that hypolimnetic alum applications may be more effective in large lakes where seiches and vertical entrainment of hypolimnetic P are more significant. The cost for these lakes could be excessive.

Details for determining and applying an alum dose are given.

This report was submitted in fulfillment of Project Number 16010 HCS, Contract Number R 801936, by Kent State University, under the (partial) sponsorship of the U.S. Environmental Protection Agency. This report covers a research period from November 1971 to January 1976, and work was completed in June 1977.

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SECTION 1

INTRODUCTION

Eutrophication is the process of enrichment of lakes with plant nutrients and particulate matter which leads to increased biological production and decreased lake volume. This is a natural process which occurs at varying rates, depending upon factors such as soil type and ratio of drainage area to lake surface area, and upon lake properties such as mean depth and flushing rate. Where activities on the land are intense, the rate may be greatly accelerated by erosion or sewage income, leading to what has been termed "cultural" eutrophication. The symptoms of eutrophication, such as excessive macrophytes, blooms of algae and fish kills may result in greatly decreased recreational, municipal, or industrial use.

Control of algal blooms must ultimately involve a reduction in external income to the lake. Experiments with algal assays (e.g., Miller et al. 1974), in situ enclosures (e.g., Powers et al. 1972), and whole lakes (e.g., Schindler, 1974) have shown that in most instances, a reduction in phosphorus concentration alone will bring about an algal bloom decrease. However, a means of macrophyte control, a more important nuisance than algae in many eutrophic lakes, is less well understood.

While there are few well-documented descriptions of the response of lakes to nutrient diversion, in those cases where diversion was sufficient to result in a large reduction in lake phosphorus concentration, the lakes have exhibited measurable improvements in quality (e.g., Edmondson, 1970) as measured by changes in plankton abundance and productivity.

Most, if not all, of the reported cases of nutrient diversion have involved the control of point sources, such as sewage outfalls. Perhaps a more common problem affecting hundreds of communities in the United States is the eutrophication of smaller, often kettletype lakes, surrounded by dense urbanization. Typically the homes are serviced by septic tanks and are located on small, completely cleared lots close to the lake shore. These lakes may have been somewhat eutrophic from their naturally rich soils, but became culturally eutrophic, with both algae and macrophyte problems, from a combination of septic tank drainage, erosion, storm run-off, and other non-point sources.

Will these lakes exhibit reduced symptoms of eutrophication if one of these several nutrient sources is diverted? One objective of this research was to address this question through long-term monitoring of the degree of

eutrophication of two small lakes after the diversion of septic tank drainage.

Several techniques have been employed to accelerate lake recovery after diversion (see review by Dunst et al. 1974). While some of these procedures are directed toward nutrient removal (e.g., flushing, harvesting) most are intended to control internal sources of nutrients. Mortimer (1971) and others have viewed the anaerobic, hypolimnetic sediments to be a potentially significant source of phosphorus. Stauffer and Lee (1973) have presented evidence to show that at least in large lakes such as Lake Mendota, thermocline erosion and co-occurring transport of P is significant in maintaining the algal blooms in the epilimnion. The second research objective was to determine whether or not the application of aluminum sulfate to the hypolimnion of one of the two lakes to control internal phosphorus loading from the sediments could accelerate the lake's recovery after septic tank diversion.

The organization of the presentation of methods and results in this report includes two principal sections. The Materials and Methods section describes the hydrological and limnological methods and a summary of our approach to aluminum sulfate dosage and application. The Results section contains the basic findings of both the immediate and longer term effects of the alum treatment and a description of the results of septic tank diversion. However, since a significant portion of this research dealt with a step-by-step development of aluminum sulfate dose, effectiveness, and tests for toxicity, a description of these findings is included as an Appendix to the hydrological-limnological findings. Thus the reader may turn promptly to the procedures, results and conclusions with a basic understanding of the aluminum sulfate methodology. Those who desire a more detailed statement of the basis for dose, or a step-by-step summary of how to ascertain dose, may refer to the Appendix and also to Kennedy (1978).

SECTION 2

CONCLUSIONS

A prompt return of groundwater, streams and lakes to near zero fecal coliform levels occurred after septic tank diversion. Leach fields, even when installed in ideal soils, did not operate properly and their use near lakes should be discouraged.

Diversion did not bring phosphorus income to an "acceptable" level (Vollenweider, 1976) over the five year post-diversion monitoring period because the lakes received untreated storm flow and run-off from diverse non-point sources typical of urban areas. Lake phosphorus concentrations remained at levels typical of eutrophic lakes. After an initial reduction in the year following diversion there was little change in the algal standing crop.

A maximum hypolimnetic dose of aluminum sulfate effectively retarded phosphorus release from lake sediments for at least one year. No adverse side effects were observed.

Net internal phsophorus loading was only partially reduced by the hypolimnetic treatment. The untreated upper zone of the lake, probably the littoral zone, was a significant internal source of phosphorus.

The alum treatment reduced epilimnetic phosphorus concentration and algal standing crop, but both remained within ranges expected in eutrophic lakes. The treatment appeared to alter the dominant bloom species from blue-green algae to co-dominance of blue-greens with dinoflagellates and diatoms.

Hypolimnetic aluminum sulfate treatment may be more effective in lakes where vertical entrainment of phosphorus-rich hypolimnetic waters is a dominant internal source of phosphorus. Such lakes are likely to be large and treatment costs may be prohibitive.

Septic tank diversion, unless accompanied by storm water control and land use regulations, will not bring about greatly lowered symptoms of eutrophication. However, diversion is essential to prevent the situation from becoming worse.

SECTION 3

RECOMMENDATIONS

Most lake restoration techniques are aimed at control of internal phosphorus loading. Contrary to expectations in this study, an important source of internal phosphorus loading appeared to be littoral in origin. These sources need to be identified and the use of aluminum sulfate to control internal phosphorus loading confined to those lakes where sediments are the primary source.

The role of groundwater in lakes should be further investigated, including studies of nutrient transport across the mud-water interface. Methods of measuring groundwater flow and knowledge of the distribution of groundwater flow over the lake bottom are especially undeveloped and require further study.

A land use plan to control non-point nutrient sources prior to lake shore development is essential. Septic tank systems, even where soils appear to be ideal, should not be used.

Where lake shore development has occurred, protection of the lakes will require, in addition to sewage diversion, a plan to prevent further development on the lake shore and to control non-point sources such as erosion and storm flows.

Macrophytes, which are at least as important as algae nuisances in the typical smaller eutrophic lake, should receive increased attention with regard to both means of control and their role in energy flow and nutrient cycling.

Continued monitoring to assess the long-term effectiveness of the hypolimnetic treatment of West Twin Lake in controlling internal phosphorus loading and reducing lake concentration is needed.

SECTION 4

WATERSHED CHARACTERISTICS

GEOLOGY

The Twin Lakes Watershed is glacial drift overlying Upper Paleozoic sandstone or shale bedrock. The flat-lying bedrock strata were deeply cut by a pre-glacial stream, and the resulting topography is a valley 100 meters deep and 2 kilometers wide, enclosed by walls of sandstone and shale. The valley has been completely filled with glacial drift. Figure 1 is a diagrammatic section of the buried valley in relation to the Twin Lakes.

The oldest bedrock is the Mississippian-Devonian shale that should be present only under the drift in the deepest part of the valley. Overlying the shale is Berea Sandstone. Interbedded shale and siltstone of the Mississippian Cuyahoga Group lies above the Berea Sandstone, and Pennsylvanian Sharon Sandstone forms the upper part of the valley walls. The valley, cut into the Mississippian and Devonian shales, was partially filled with Pleistocene Illinoian deposits comprised of gravel and some finer grained material. This gravel unit is the principal aquifer for the wells of the homeowners in the Twin Lakes Watershed.

Overlying the Illinoian gravels is the coarse, sandy and pebbly gray Mogadore Till (Winslow and White, 1966). The Kent Till overlying the Mogadore is a sandy, silty, calcareous till, and is the surface material on the southern and eastern part of the watershed. On the north and west, the Kent Till is overlain by Hiram Till, a gray silty clay.

The Twin Lakes Watershed lies within the western belt of the Kent End Moraine of early Wisconsin age. Wasting of the stagnant ice resulted in the formation of kettle depressions separated by outwash deposits of sand and gravel. The kettles became sinks for fine grained material from the wasting ice and thus developed partial bottom seals. The Twin Lakes are formed in these Kent Till depressions. The surficial geology of the watershed is shown in Figure 2.

The principal soil types are those formed on upland clay till (Ravenna), upland gravel and sandy till (Chili), lowland silt and clay (Orrville), and lowland peat and muck (Carlisle). These four major soil types differ in their water holding and transmitting capacities, and potential for erosion. The Ravenna soils on the clay till north of West Twin Lake are susceptible to erosion and have low infiltration capacity and low groundwater recharge.

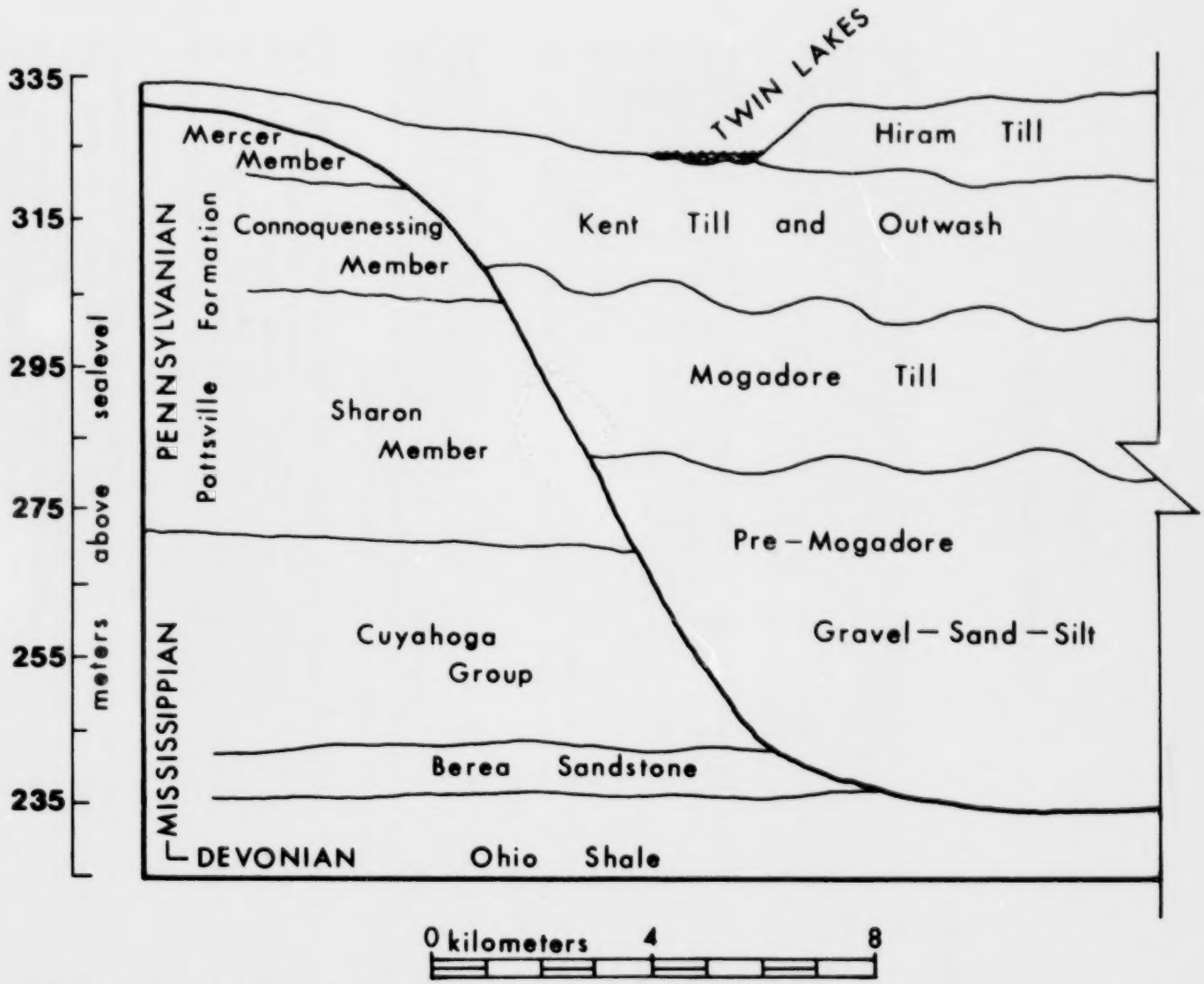

FIGURE 1. Glacial stratigraphy of the Twin Lakes Watershed

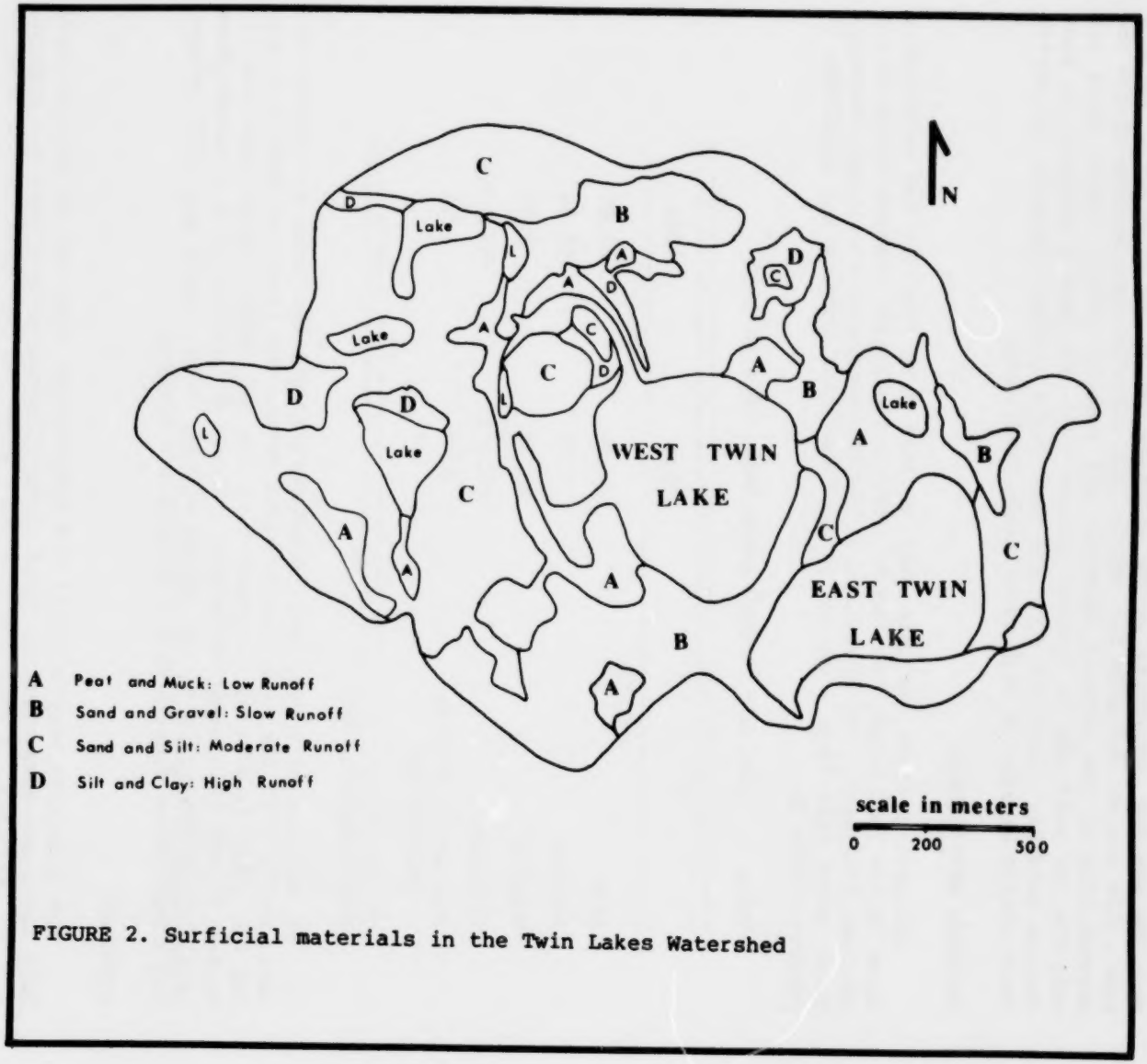

FIGURE 2. Surficial materials in the Twin Lakes Watershed

Chili soils on the south of West Twin and around the edges of East Twin are less susceptible to erosion and have a high infiltration capacity.

The relationship of the soils to septic tank installation and subsequent problems was not considered in the urbanization of the watershed. Ravenna type soils are not suitable for septic tank installations because of the low permeability of the subsoil (Ritchie and Powell, 1973). Chili soils are suitable for most septic systems because they may permit effluent to pass through subsoil. Carlisle muck and peat soils are not suitable for septic leach fields because of high water table conditions.

CULTURAL DEVELOPMENT

Earliest records indicate that by 1803 five families had established farms in the Twin Lakes area (Brown, 1885). Population changed little during the next 100 years. Tax maps filed at the Portage County Courthouse (Ravenna, Ohio) since 1850 show a change in land use from farming and woodlands to pasture (Table 1).

Table 1. LAND USE IN TWIN LAKES AREA, 1850 to 1880

	1850	1880
Plow	40.0%	32.0%
Pasture	8.6%	55.0%
Woodlot	51.4%	12.5%
Land used	45.3 ha	266.4 ha
Population density (persons/ha)	0.57	1.90

From Otterson, 1974

At the turn of the century, developers bought the farms surrounding the lakes and by 1918 the major subdivisions around the lakes had been laid out, and the shift from rural to residential use had begun. Today there are 374 houses on the Twin Lakes Watershed. Using an average of 4.2 persons/home (Statistical Abstracts of the United States, 1973), there are 1,571 persons and a population density of 6.2 persons per hectare.

LIMNOLOGY

The principal morphological and hydrological features of East and West Twin Lakes are summarized in Table 2 and illustrated in Figure 3. Averages are for 1972-1976. A more complete description has been presented in Cooke et al. (1973) and Waller et al. (1976).

Table 2. LIMNOLOGICAL FEATURES OF THE TWIN LAKES WATERSHED

Latitude - Longitude	$41^{\circ}12'$ North; $81^{\circ}21'$ West	
Area of watershed (ha)	334.5 (including lakes)	
	West Twin Lake	East Twin Lake
Lake area (ha)	34.02 (30.22*)	26.88
Maximum length (km)	0.65	0.85
Maximum width (km)	0.60	0.50
Volume (M^3)	14.99×10^5	13.50×10^5
Maximum depth (M)	11.50	12.00
Mean depth (M)	4.34 (4.45*)	5.03
Elevation (M)	318.73	318.42
Subwatershed area (ha)	233.02	101.48
Area of other lakes in subwatershed (ha)	15.00	3.00
Mean water residence (yr)	1.28	0.58
Mean areal water load (M/yr)	3.50	9.12
Subwatershed population	1124	1571**

*Shallow canals are contiguous with West Lake, but are excluded as part of the lake in averaging some limnological features.

**Includes the population of the West Twin subwatershed since this lake drains into East Twin.

Both lakes are dimictic, second class lakes (defined by Hutchinson, 1957). In this report, seasons were based on water temperature as follows: winter (4° C), spring (unstratified), summer (metalimnion present), and fall (unstratified). Ice forms in December and remains until early March. The 1971-76 average onset of summer stratification was 15 April for both lakes, and the average summer lasted 210 days in East Twin, 198 days in West Twin. By mid-November, both lakes had circulated.

Dissolved oxygen is rapidly depleted from deep waters following the onset of summer stratification and partially depleted after ice formation. Associated with thermal and dissolved oxygen stratification is a marked stratification of phosphorus and nitrogen forms. The annual ranges of nitrogen, alkalinity, pH, conductivity, and transparency are listed in Table 3. Changes in phosphorus concentration with depth and over the seasons are described in the Results section.

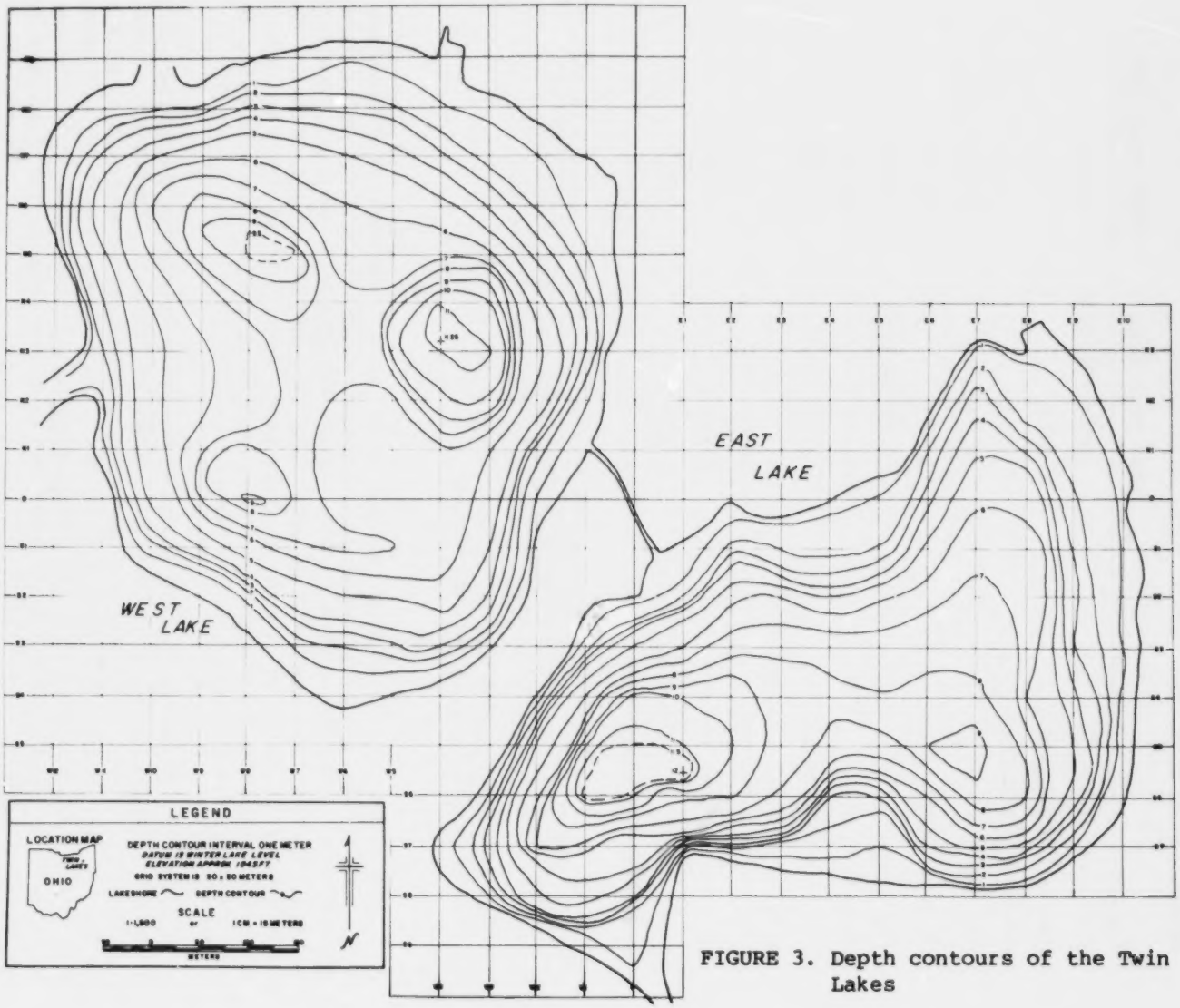

FIGURE 3. Depth contours of the Twin Lakes

Table 3. SEASONAL AND ANNUAL VOLUME-WEIGHTED AVERAGES (1971) OF SELECTED CHEMICAL AND PHYSICAL FEATURES OF THE TWIN LAKES (µg/l)

	Winter	Spring	Summer	Epilim	Hypolim	Fall	Year
1. Ammonia (NH_4-N)							
West Twin	293.4	223.2	745.6	387.3	3288.5	1336.0	673.0
East Twin	296.2	99.3	662.3	318.9	3190.5	924.9	582.2
2. Nitrate-nitrite (NO_3-N)							
West Twin	268.2	100.9	56.8	36.1	80.9	101.4	125.7
East Twin	129.1	69.9	65.7	43.2	102.0	95.6	89.1
3. Conductance (µmho)							
West Twin	411.8	411.8	401.2	393.0	435.0	385.3	402.3
East Twin	375.7	375.7	361.4	352.4	398.0	370.7	367.2
4. pH* (1973)							
West Twin	7.92	7.91	7.58	7.76	7.30	7.67	7.68
East Twin	7.54	7.54	7.53	7.94	7.23	6.81	7.38
*log of mean anti-log							
5. Alkalinity (mg $CaCO_3$/l)							
West Twin	109.4	109.4	107.9	101.9	144.5	115.7	109.9
East Twin	104.3	104.3	104.9	95.0	146.2	105.8	105.0

Characteristics of the littoral, sub-littoral, and profundal surficial (upper 2 cm) sediments were studied by Lardis (1973). Littoral sediments contain mainly decaying vegetation, shell fragments, and allochthonous debris. Sublittoral sediments are brown to black-gray with lesser and varying amounts of decaying vegetation. Profundal muds are black and very fluid. The organic content of dried sediment samples ranges from 14% in the littoral to about 40% in the profundal. The organic phosphorus content of the muds increases with depth of overlying water and is significantly ($p < .05$) less in ETL (mean=66 µg P/g) than in WTL (mean=85 µg P/g). The phosphorus content of profundal muds declines after the onset of thermal stratification.

The flora of the lakes are dominated by a large macrophyte community which covers about 25% of the lake area (Rogers, 1974) and by a succession of phytoplankton blooms. Potamogeton crispus is the dominant macrophyte in April, May, and June; late summer growth is dominated by dense populations of Elodea canadensis, Chara vulgaris, Najas flexilis, and Ceratophyllum demersum. After ice-out a bloom of diatoms, usually Fragilaria crotenensis and Asterionella formosa, occurs, followed by an abrupt change in April or May when high water clarity and low plankton density is typical. Oscillatoria rubescens, common in the spring, is the dominant species of the metalimnion. In the summer a series of blue-green algae blooms occurs, culminating in a heavy bloom of Aphanizominon flos-aquae or Anabaena limnetica in August and September.

The zooplankton have been studied by Heinz (1971). Large populations are found in spring and fall, dominated by Daphnia galeata mendotae and Cyclops bicuspidatus thomasae. The bottom fauna have not been examined quantitatively, but appear to be dominated by Chaoborus. The fish community is composed largely of members of the Centrarchidae.

SECTION 5

MATERIALS AND METHODS

HYDROLOGY

The equation for a hydrologic equilibrium was used to measure the water balance of the lakes from 1972 through 1976. It was assumed that the following equation must balance each year:

inflow + precipitation = outflow + change in storage.

Inflow was incoming streams + groundwater + overland flow, and outflow was outgoing streams + evaporation + groundwater.

Precipitation was collected at a recording weighing bucket rain gauge located on the north side of the watershed. The location of this station is shown on Figure 4. Data were also collected from nearby (1 km east) Lake Rockwell and from Kent (8 km south). The normal or 30-year average precipitation was taken from the Hiram station approximately 29 km north of Twin Lakes. For 1974-76 precipitation at Twin Lakes was higher than normal.

Evaporation was estimated in 1972 using the Blaney-Criddle equation and temperature from the Hiram station. In 1973 and 1974 evaporation was measured with a recording evaporimeter and a Class A evaporating pan. The evaporimeter was placed near the lake for the spring, summer, and fall months. No data were gathered during periods of freezing temperatures. The Class A pan was installed on the roof of a low building in Kent. For the years 1975 and 1976 evaporation data from Coshocton, Ohio, 160 km south of the watershed, were used.

The overland flow from individual precipitation events was computed using the U.S. Soil Conservation Service technique described by Mockus (1969). This technique is based on soil infiltration characteristics, 5-day antecedent precipitation, and land use. No actual measurements of overland flow were made.

Surface flow into West Twin, East Twin, and Dollar Lakes (Figure 4) was measured at 14 stations. Flow at six stations was perennial and was continuously monitored at three locations. Table 4 gives the stream number, type of flow, and measuring technique. The technique employed was determined by the flow regime of the stream and the stream gradient. Stream 8S, a perennial stream discharging from West Twin Lake was not monitored

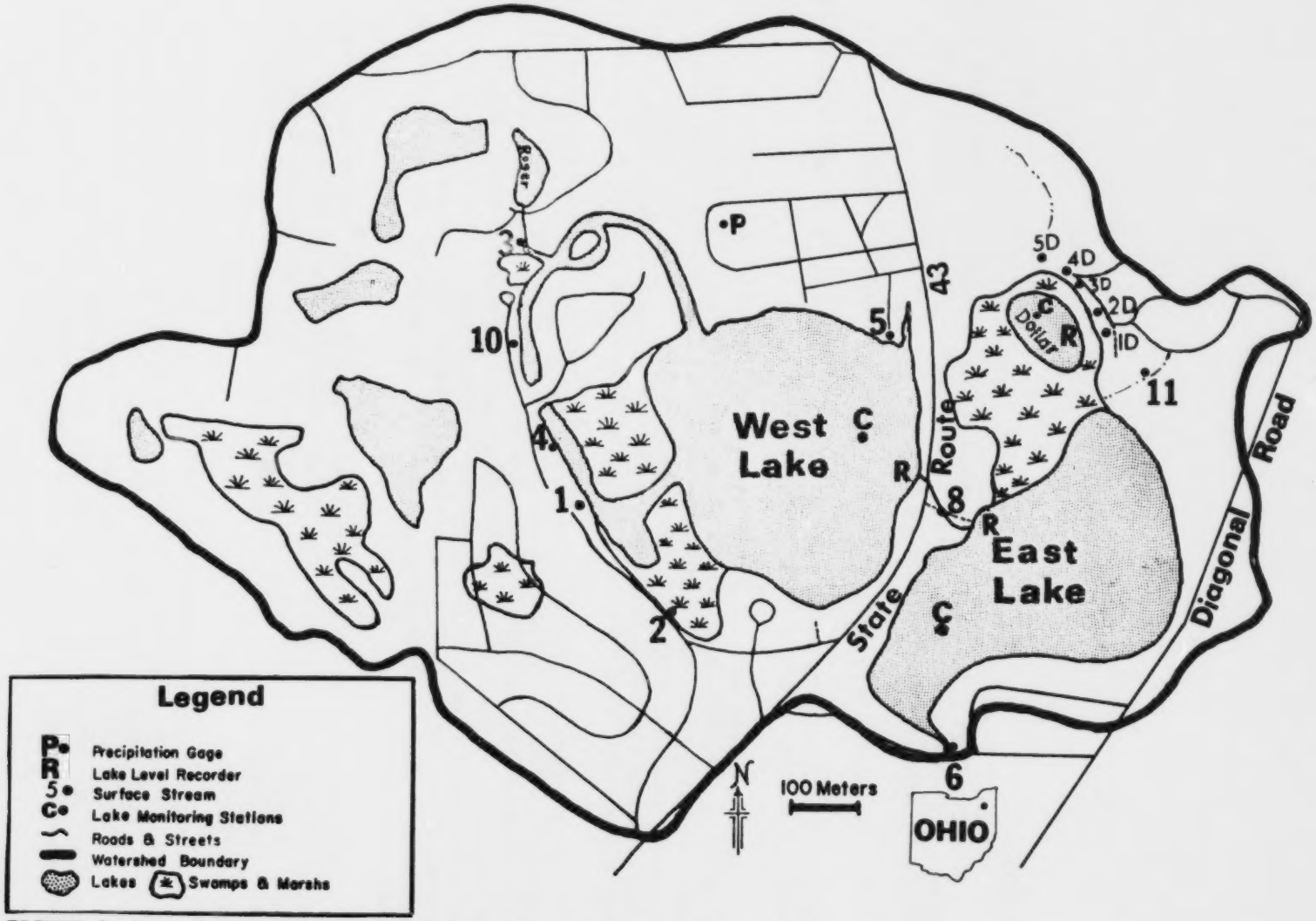

FIGURE 4. Location of streams entering Twin Lakes and Dollar Lake

Table 4. SOURCE, RECEIVING WATER AND MEASURING TECHNIQUES OF SURFACE STREAMS IN THE TWIN LAKES WATERSHED.

Stream No.	Source	Receiving Water	Flow	Measurement Method
1S	Pond overflow and storm drain	West Twin	Perennial	90° V-notch weir, stilling well and recorder
2S	Street drain and depression	West Twin	Perennial	90° V-notch weir, stilling well and recorder.
3S	Upland lake overflow	West Twin	Dry in summer	H flume and stilling well with recorder.
4S	Lake overflow and street drain	West Twin	In response to storms	Culvert discharge
5S	Storm water in urban area	West Twin	In response to storms	Culvert discharge
6S	East Twin Lake	Cuyahoga River out of watershed	Perennial	Stage recorder in stilling well.
7S (4D)	Stormwater	Dollar Lake	In Response to storms	Culvert discharge
8S	West Twin Lake	East Twin Lake	Irregular in summer	Current meter and stage recorder 1972-1973
10S	House foundation drain	West Twin	Perennial	Culvert discharge
11S	Storm flow wooded area	East Twin	In response to storms	90° V-notch weir & bucket
12S (2D)	House foundation & storm drain	Dollar Lake	Perennial	Culvert discharge
1D	Foundation and lawn drainage	Dollar Lake	In response to storms	Culvert discharge
3D	Storm flow urban area	Dollar Lake	In response to storms	60° V-notch weir & stilling basin
5D	Storm flow-urban area	Dollar Lake	In response to storms	90° V-notch weir & measuring bucket.

continuously because of low gradient. This stream connects West Twin and East Twin Lake, and on occasion during the dry summer months the flow reversed so that East Twin Lake flowed into West Twin Lake. No structure was built on stream 5S because of flooding potential in and around this shallow channel low gradient stream. Flow into Dollar Lake was monitored because this small lake is separated only by a Sphagnum marsh from East Twin and thus is part of East's water budget.

Five streams, 1S, 2S, 3S, 4S, and 5S enter West Twin Lake. Three (1S, 2S, and 3S), which account for 50-75% of the annual stream income, were measured continuously or at a minimum, weekly. Two streams, 4S, an upland man-made lake outlet, and 5S, an urban stormflow channel, were measured infrequently in 1972 and not at all in 1973 and 1974. Their 1972-74 flows were estimated by regression equations based on data from 1975. The estimate of 4S outflow in 1972-74 was obtained by a seasonal linear regression of its daily 1975 outflow to that of daily outflow from stream 3S. Both streams are upland lake outflows which discharge into West Twin. The correlation coefficients were: summer $r = 0.600$, fall $r = 0.738$, and spring $>$ 1000 M^3 /day, $r = 0.690$; and spring $\leq$ 1000 M^3 /day $r = 0.805$. Winter outflows of 4S were assumed to be a constant proportion of the outflow of 3S, based on the 1975 ratio of winter outflows. Stream 5S, the urban drainage, was estimated by a log-log regression of its 1975 monthly flow against the 1975 monthly flow of all other streams entering West Twin ($r = 0.681$ for all seasons). In 1975 the rate of flow of all streams in the watershed was measured continuously or daily.

Change in lake water levels was measured by a water level recorder and the data used to estimate change in storage.

To determine groundwater contribution to the Twin Lakes, a combination of lakeside wells and in-lake piezometers was installed. In 1971 16 wells were drilled in the watershed to determine the elevation of the water table. These wells established that groundwater flow was towards the lakes. Hydraulic conductivities were determined on each well using the slug injection method (Ferris et al., 1962). In 1972 and 1973 an additional 20 wells were drilled on the periphery of the lakes. Stratigraphy of the near shore wells showed an upper layer of saturated sand overlying a silt and clay layer. Shallow groundwater flow to the lakes was assumed to be the water passing through the upper sand layer into the lake. The hydraulic gradient was determined from the water table elevations at the phreatic divide and the elevation of the lake divided by the distance from the lake to the phreatic divide. Each lake was divided into 16 "pie" slices (Cooke et al. 1973), and total shallow flow was the sum of the flow across each of the slices. Measurements of the water table levels at the divide and at the lake showed only minor monthly fluctuations so the hydraulic gradients for each slice were assumed constant. Hydraulic conductivities for the tested wells varied by less than one third of an order of magnitude, so a constant value for hydraulic conductivity was used. The seepage value determined for West Twin Lake was 196 M^3 /day and 137 M^3 /day for East Twin Lake.

Deep groundwater flow was computed by installing two deep piezometers (13 and 6 meters deep) into the aquifer beneath the confining silt layer. Using the hydraulic gradient determined from these two wells as a vertical gradient, and the hydraulic conductivities of the surface, seepage flow was computed for the area of the lake basins. These values were 646 M^3 /day for West Twin and 535 M^3 /day for East Twin. These values were used in water budget computations in the first published reports from this project (Cooke et al. 1973).

Szczepanowski (1976) attempted to verify the flow rates into East Twin obtained from lakeside wells by installing piezometers in the lake. Thirty-two piezometers in 13 nests were installed to depths ranging from 0.63 to 17.5 meters below the water surface. The deepest penetration into the lake mud was 8.5 meters. His data gave an average inflow of 207.1 M^3 /day for East Twin Lake from both shallow and deep systems, a value 3.2 times lower than that computed using the flow net technique. His rate was believed to be too low because deep spring flow was not measured.

Analysis of the water budget and the lake outflow of East Twin suggested that the values of both 672 M^3 /day (137+535 M^3 /day) and 207.1 M^3 /day were probably too low; a high residual inflow value could not be accounted for. In 1975 daily stream monitoring and careful analysis of storm flows, precipitation and evaporation still did not eliminate a high residual.

Over the five-year monitoring period groundwater inflow was thus estimated by the use of lakewide wells (Buller, 1974), piezometers (Otterson, 1974; Szczepanowski, 1976), by the use of thermal gradient and seepage meters (Cooke et al. 1975), and by lake water balance residuals (Cooke et al. 1977). A comparison of these results is given in Table 5. No single technique proved to be entirely satisfactory in determining the groundwater inflow component, and the rates obtained varied by over an order of magnitude.

Table 5. RATE OF ANNUAL GROUNDWATER FLOW INTO THE TWIN LAKES, COMPUTED FROM FIVE TECHNIQUES

Method	Rate (M^3/yr)
Lakewide wells and flow net	2.5×10^5
Piezometers in the lake	7.6×10^4
Thermal gradient	3.6×10^6
Seepage meters	9.1×10^5
Water balance residual	1.1×10^5 to 7.0×10^5

In contrast to all earlier reports of the water-phosphorus budget of the Twin Lakes, the monthly groundwater inflow in this report was obtained by assigning all residual water income from the equation for a hydrologic equilibrium to groundwater. Measured groundwater flow rates were not used. If the value was negative, it was assumed that water was lost from the lakes via groundwater. This technique was chosen since much of the residual was probably interflow which was not measured, because there are numerous deep springs also not measured, and because accurate measures of surface flows, particularly in 1975, were obtained. When an alternative method was employed, as was done in Cooke et al. (1973), Cooke and McComas (1974), Cooke et al. (1975), and Waller et al. (1976), then the monthly residual was assigned to overland flow, giving this component of the equation an unrealistically high flow.

The methods employed for chemical analysis of incoming water (phosphorus, coliform bacteria) are described in the Limnology section.

LIMNOLOGY

Each lake was sampled weekly over the deepest part (Figure 3), from the time of ice-out in the early spring until ice formation, and then weekly through the winter as permitted by ice conditions.

Temperature was measured to the nearest 0.5° C at one-meter intervals using a YSI resistance thermometer. Transparency was determined with a 20 cm diameter Secchi disc, with black and white alternating quadrats, and recorded to the nearest 0.1 M, averaged from three readings. Lake water was collected at one-meter intervals with a 1.9 l Kemmerer bottle and placed into 300 ml glass-stoppered BOD bottles for determination of dissolved oxygen concentration by the azide modification of the Winkler method (APHA 1965).

Lake water samples were also collected at 1 M intervals, placed in acid-rinsed polyethylene bottles, and returned to the laboratory for physical, chemical, or biological determinations. The following determinations were performed on whole lake water: conductivity, pH, alkalinity, total phosphorus and alkaline phosphatase. The following were performed on water filtered through a Whatman GF/A glass fiber filter: total phosphorus (to yield a "filterable total phosphorus" (FTP) estimation) sulfate, and aluminum.

Conductivity was determined with a YSI Model 31 conductivity bridge, pH with a Corning Model 7 pH meter, and alkalinity by titration of whole lake water with 0.02 N $H_2 SO_4$ to an endpoint of pH 4.5 (APHA, 1965). Soluble reactive phosphorus (SRP) was determined by the Murphy and Riley (1962) procedure on 0.45 µm Millipore filtered water (USEPA, 1971). Total phosphorus (TP) was determined by the above colorimetric procedure, following persulfate-sulfuric acid digestion of unfiltered water (USEPA, 1971). Filterable unreactive phosphorus (FUP) was calculated as FTP-SRP. Alkaline phosphatase activity was determined by incubating 2.5 ml of whole lake water with 0.3 ml (0.1 M tris, .01 M $MgCl_2$ pH 9.0) and with 0.3 ml (0.3 mg p - nitrophenyl phosphate/ml). Activity was determined as the rate of increase of

absorbance at 395 nm over four hours, measured in 1 cm cuvettes (Heath and Cooke, 1975). Sulfate was determined turbidimetrically (Hach Chemical Co.), and aluminum colorimetrically (Hach Chemical Co.). On these tests the results were compared to our standard curves and not those provided by Hach Chemical Company.

Total seston was determined gravimetrically by filtering 500 ml of lake water onto tared Whatman GF/A glass filters, air dried at 98° C, and weighed to the nearest 0.1 mg without ashing (USEPA, 1971).

Chlorophyll a was determined by filtering 500 ml of lake water onto Whatman GF/A glass filters and extracting with 90 percent Mg CO_3 - buffered acetone in a tissue grinder (Long and Cooke, 1971). The trichromatic method of Strickland and Parsons (1968) was used without acid correction. Often samples were filtered and stored frozen before grinding and extraction.

Phytoplankton cell volume was measured by filtering 25 ml of lake water onto 0.45 µm Millipore (HAWP) filters. These were air dried, cleared with immersion oil, and cells counted at a magnification of 140X. Eleven Whipple fields were counted. Approximate geometric shapes were used to estimate the cell volume (McNabb, 1960).

Coliform bacteria were determined by filtering 100 ml of lake water onto gridded 2.5 cm Millipore 0.45 µm filters. These were incubated on sterile EMB lactose agar at 30° C for 24 hours. Coliforms were identified as the "metallic" colonies (APHA, 1965).

ALUMINUM SULFATE APPLICATION

Although aluminum sulfate (alum) applications to eutrophic lakes have been attempted (e.g., J.O. Peterson et al. 1973), additional studies of effectiveness and longevity were needed, and the dosage guidelines also needed evaluation. Our objectives were to establish dose determination methods, develop application methods, and then evaluate effectiveness, toxicity, and longevity through laboratory and field experiments.

Aluminum sulfate has been used in the clarification of water supplies and in the removal of phosphates from wastewater effluents. Dosage is commonly determined in "jar-tests" in which stepwise additions of alum are made to determine the minimum dose at which maximal removal of either suspended solids or phosphorus can be achieved. Effective dosages used for the removal of phosphorus from wastewater are expressed as aluminum to phosphorus molar ratios since theoretically one mole of aluminum is required to precipitate one mole of phosphorus as $AlPO_4$ (Stumm, 1964). In practice molar ratios of 1.5 to 2.0 are common (Nesbitt, 1969).

S.A. Peterson et al. (1974) suggested that the calculation of aluminum sulfate dosage for lake treatments could be based on molar ratios and reported maximal phosphorus removal (95%) from algal assay medium and pond water at molar ratios of 3.0, 4.0 respectively. Such a method places emphasis on the

removal of phosphorus from the water column. However, the success of any lake treatment will depend on the amount of alum added and the ability of the added alum to reduce the release of phosphorus from anoxic sediments, and only secondarily on the removal of phosphorus from the water column at the time of application. Accordingly, an alternative approach to dose calculation was sought.

The reaction of aluminum with carbonate alkalinity results in the formation of aluminum hydroxide. Since most eutrophic lakes are characteristically alkaline, dose can be based on this reaction:

$$Al_2(SO_4)_3 + 3Ca(HCO_3)_2 \ddagger 2Al(OH)_3 + 3CaSO_4 + 6CO_2$$

A dose based on alkalinity and not phosphorus concentration allows addition of aluminum sulfate up to a point at which most of the alkalinity is removed, pH values fall, and dissolved aluminum concentration begins to increase. Phosphorus in the water would be removed by entrapment of particulate matter in the polymerized amorphous aluminum hydroxide floc, and by adsorption of dissolved inorganic phosphorus to charged sites on its surface (Lea et al. 1954) during sedimentation. Depending upon the amount of phosphorus in the water column prior to treatment and upon the removal efficiency of the hydroxide, the resulting floc layer covering the sediments could continue to be effective by retaining phosphorus leaving sediments and thus produce a "chemical barrier" over anoxic sediments. Directed towards the reservoir of phosphorus in sediments rather than towards that in the water column at the time of application, this approach emphasizes the possibility of a long term means of improving water quality by control of internal loading or release of phosphorus from sediments.

A step-wise experimental program to substantiate this approach to dose determination involved laboratory studies to evaluate floc formation and phosphorus removal characteristics, controlled in-situ enclosure experiments, laboratory and field toxicity sutdies, and the full-scale pilot treatment of a small lake (Dollar Lake). These experiments are outlined in the Appendix and described in detail in Wilbur (1974) and Kennedy (1978).

It was found that maximum aluminum dose, defined as that dose above which residual dissolved aluminum concentration exceeds 0.05 mg Al/l (a value which Everhart and Freeman (1973) report can be tolerated by trout), was linearly related to alkalinity and could be determined by simple jar tests. Dose determination for West Twin Lake is described in the Appendix, Section G.

The treatment of phosphorus-spiked lake water samples with a maximum aluminum dose and the elution of the resultant floc material with phosphate buffers (pH 6.0, 7.0, 8.0) indicated that phosphorus removal efficiency was high (96%) even at high phosphorus concentrations (1150 µg P/l) and that the floc retained the capacity for continued phosphorus adsorption following sedimentation from the water column. Controlled field testing with columns of lake water (Kennedy, 1978) and field and laboratory toxicity tests (Wilbur, 1974) confirmed that long term (104 days) control of phosphorus release from

sediments could be achieved with an alkalinity-dictated maximum dose without adverse ecological impacts. The decision to treat hypolimnetically was based on results of preliminary enclosure experiments (Kennedy, 1978) in which surface applications of alum resulted in decreased removal efficiency in deeper, phosphorus-rich hypolimnetic waters, and to the desire to avoid the possibility of localized toxicity problems in the epilimnion due to decreased pH.

Based on an evaluation of the results of these studies, the decision was made to treat West Twin Lake, maintaining East Twin Lake (100 M downstream) as a diversion-only control. East Twin is not the perfect control since it receives over half its water from West Twin and, therefore, should ultimately benefit from a reduction of phosphorus concentration in West Twin. Dose and application methods developed during the pilot treatment of Dollar Lake were used. The lateral extent of the upper boundary of the hypolimnion was marked at the surface with buoys and the entire enclosed area divided into treatment quadrats of 100 x 50 M. Measurement of the volume of the hypolimnetic portion of each quadrat and their alkalinities allowed the calculation of the maximum amount of alum to be applied to each. The application of the 139.55 M^3 (36,919 gals) of aluminum sulfate just above the hypolimnion (5 M) required two specially designed barges working 18 hours/day for a total of three days (July 29-31, 1975). A detailed description of dose determination calculations and logistics, including storage and delivery equipment, can be found in the Appendix. Also included is a cost analysis.

SECTION 6

RESULTS

INTRODUCTION

The Results section has been organized into five parts. First, since it was the presumption that septic tank diversion could lower nutrient concentration in the lakes, and that control of in-lake phosphorus loading through the aluminum sulfate application could lower it further, evidence is presented here that the phytoplankton of the Twin Lakes were phosphorus-limited and therefore could respond to a significant decrease in concentration. The most immediate effect of diversion should be on the water-nutrient budget. This section is presented next. The phosphorus content of the lake may be altered by a change in both external and internal income, and so the effect of diversion on lake concentration, and then the immediate and longer-term effects of the alum treatment are described. The response of the biota to diversion and alum treatment is given next, and then a final section quantifying the effects of septic tank diversion and alum treatment in a Trophic State Index is presented.

EVIDENCE OF PHOSPHORUS LIMITATION

It has been shown that the growth of phytoplankton in freshwater ecosystems is often, but not always, limited by the availability of phosphorus. The limitation of phytoplankton growth can be detected at the ecosystem, the community, the population, and the biochemical levels of organization. Here evidence is provided from each of these levels consistent with the view that the phytoplankton of the Twin Lakes were phosphorus-limited.

In comparison with the other micronutrients, phosphorus is among the least abundant in lake ecosystems. Often during the growth of the planktonic community the available soluble reactive phosphate is driven to barely detectable levels, a necessary though not sufficient condition to establish phosphorus limitation. Throughout the duration of this project, particularly in East Twin, it was consistently noted that orthophosphate (measured as soluble reactive phosphorus) reached extremely low levels (less than 5 µg P/l) early in the growing season; this low concentration persisted throughout the growing season.

Chiaudani and Vighi (1974) investigated the maximum cell density attained (cells/vol) by Selenastrum capricornutum grown in filtered natural lakewater with additions of NO_3-N and or PO_4-P. When the water contained an initial weight ratio of inorganic N to soluble reactive phosphorus (SRP)

greater than 10, maximum potential algal cell density was phosphorus limited, but with an initial weight ratio (N:P) less than 5, maximum potential cell density was nitrogen limited. Determination of inorganic N (NH_4 + NO_3+ NO_2) and SRP were used to examine the Twin Lakes N:P ratio according to this view. Table 6 provides values for 1972 and 1973 (the only years in which NO_3 was determined). Both lakes in each year appeared to be phosphorus-limited lakes from these considerations. This conclusion must be regarded cautiously because it is not certain that this laboratory method would yield similar results in the field and for species other than Selenastrum capricornutum. Schindler (1977) recently suggested that phytoplankton dynamics are not as simple as the view that Chiaudani and Vighi propose. Schindler points out that decreasing the N:P ratio in lakes may exacerbate the problem by stimulating blooms of nuisance N-fixing blue-green algae.

Table 6. INORGANIC N/SOLUBLE REACTIVE P WEIGHT RATIOS

	WEST TWIN LAKE				
	Annual	Winter	Spring	Epilimnion	Fall
1972	6.00	3.80	17.65	29.70	7.38
1973	6.31	6.90	8.12	10.11	6.37
	EAST TWIN LAKE				
	Annual	Winter	Spring	Epilimnion	Fall
1972	9.51	11.22	3.84	18.49	9.98
1973	10.91	12.75	20.18	18.63	14.04

Dillon and Rigler (1974a) noted a strong correlation between the total phosphorus concentration during spring overturn and the mean chlorophyll a concentration in summer epilimnetic waters. That is, the summertime content of chlorophyll a in the lake (and by inference the summertime primary production in the lake) is related to (i.e., is limited by) the phosphorus available at the start of the growing season. If the mean summer chlorophyll concentration is related to the springtime total phosphorus content of the lake according to the relationship of Dillon and Rigler

$$\log \text{ (Chl a)} = 1.583 \log (P_T) - 1.134$$

$$r = .975$$

it may be inferred that the production of chlorophyll was continued by availability of phosphorus. Figure 5 represents the East and West Twin positions relative to the regression. All values but West Twin in 1972 and 1973 fall within the 95% confidence interval given by Dillon and Rigler. As

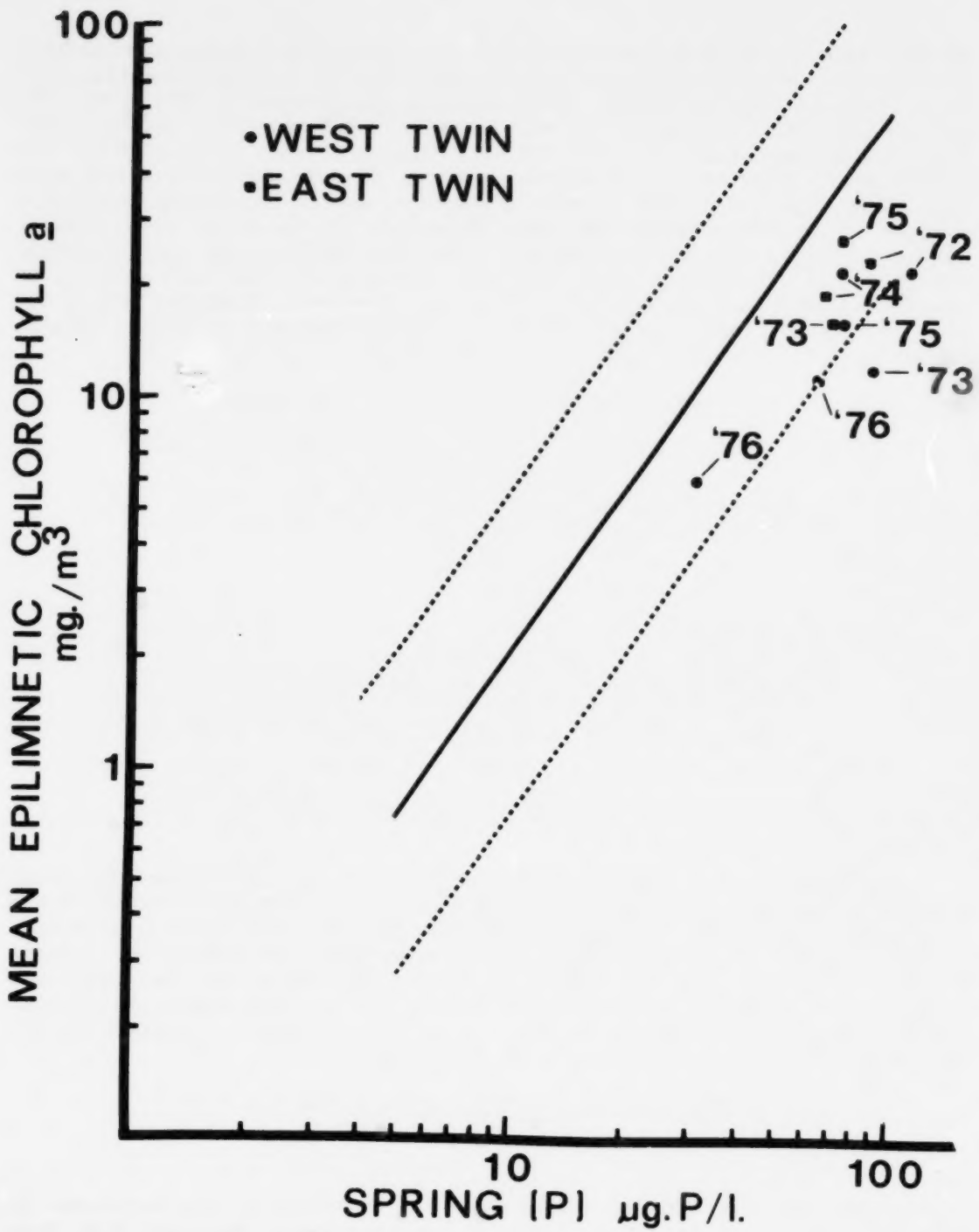

FIGURE 5. Relation between mean epilimnetic chlorophyll a and spring phosphorus concentration at Twin Lakes. Solid line is line of Dillon and Rigler (1974a). Dashed lines are 95% confidence limits.

noted in a later section on phytoplankton responses to diversion and alum treatment, 1973 was a year of heavy algicide application to West Twin Lake.

The effect of additions of phosphorus to lakewater on the growth of individual populations growing in that water may be used as an indication of phosphorus limitation. If the populations are phosphorus-limited, addition of phosphorus and only phosphorus would be expected to yield a corresponding response in the growth of the population. This approach has been refined by the use of certain "indicator species" in the Algal Assay Procedure (USEPA, 1971). A modification of the Bottle Test (Long, 1977) was used. An initial innoculation of $4X10^4$ cells/ml^{-1} of Selenastrum capricornutum Prinz was added to replicate samples of lakewater from East Twin which had been filtered through 0.45 µg membrane filters. Three bottles were held as controls. Additions of phosphorus were made to the other bottles so that the final concentration was 100 µg P/l. The samples were incubated under the prescribed temperature and light conditions and shaken continuously for 10 days. At the end of this time the cell density was determined with a Coulter Counter, Model B-. The significant differences between the spiked and unspiked samples, as judged by student's "T" test, indicated that the water was phosphorus-limiting at least for the test species (Table 7).

Table 7. ALGAL ASSAY PROCEDURE: EAST TWIN LAKE

Date	Filtered ETL Water	Filtered + Spike 100 µg P/l
27 Mar 76	36.4 ± 4.7	119.9 ± 68.2
5 Apr 76	32.5 ± 3.0	581.3 ± 9.6
12 Apr 76	41.4 ± .8	552.0 ± 30.4
19 Apr 76	56.5 ± 1.1	604.7 ± 24.3
26 Apr 76	38.9 ± 2.5	479.3 ± 21.9
3 May 76	99.6 ± 34.4	359.7 ± 24.7
12 May 76	41.7 ± 2.1	263.0 ± 68.0
19 May 76	229.0 ±164.4	524.0 ± 64.3

From Edwin T. Long 1977.
(mean cells/ml) ± 2 (std error)

It would be naive to suppose that algal populations responded only to available nutrients and therefore it is difficult to extend these results to predictions of field behavior. However, they do indicate that of the available nutrients, phosphorus was likely the limiting nutrient, at least at the time and place of the test (March - June, 1976).

Algal cells respond to phosphorus limitation by depleting excess stores of polyphosphates (so-called luxury storage compounds) and by adaptively producing acid or alkaline phosphatase. Furthermore, Jones (1972) has demonstrated a positive correlation between the degree of eutrophication and alkaline phosphatase activity, indicating that phosphatase-producing phytoplankton and bacterioplankton may be characteristic components of eutrophic

systems. Not all species are capable of this adaptive response (Kuenzler and Perras, 1968), but those which are may be identified as growing under phosphorus-limiting conditions by the appearance of enzyme activity. Fitzgerald and Nelson (1966) suggest this as a general procedure for identification of phosphorus limitation.

This view is complicated by the finding that some bacterioplankton and phytoplankton constitutively produce alkaline phosphatase, and a number of zooplankton also may continuously release phosphatases directly into the water. Therefore detection of phosphatase activity alone is not sufficient to document phosphorus limitation of the species producing it. It must be shown that the species producing it does so adaptively (i.e., that the cellular specific activity of the enzyme is variable and a function of available phosphorus).

Both East and West Twin were monitored for alkaline phosphatase activity at weekly intervals from 1972 through the summer of 1975; water was drawn from the 0.1, 2, 4, 7, and 10 M strata. Activity was consistently greatest in the epilimnion, least in the hypolimnion. Also, the majority of the activity was particulate (i.e., retained on filters) rather than soluble. Passage of whole lakewater through filters of differing porosity indicated that enzyme activity was filtered from the water coincident with the removal of phytoplankton, indicating that little if any of the activity was associated with unattached bacterioplankton or was dissolved in the water (Figure 6).

The results are graphed as activities (µmoles hydrolyzed/hr/l whole lakewater) and as specific activities (activity µl/l cell volume of species producing the enzyme) with respect to time (Figure 7). Only the results obtained from the 0.1 M stratum are presented. The greatest activities and the greatest specific activities occurred late in the growing season, and maximum activities were always coincident with phytoplankton blooms, especially blooms of Aphanizomenon flos-aquae. Previous studies have shown that this species is capable of adaptive production of alkaline phosphatase when grown under phosphorus limitation in laboratory cultures (Heath and Cooke, 1975).

Because the activity was predominantly associated with phytoplankton shown to be capable of adaptive production of this enzyme under phosphorus limiting conditions, and because the specific activity was variable and increasing with time, it can be concluded that at least those species responding in this manner were P-limited in their growth.

Eutrophication is a process occurring at the ecosystem level of organization, but its effects may be observed at lower levels. Not all studies reported here were carried out for the duration of the project and one (algal assay) was performed only on East Twin. Even so, the investigations at the ecosystem (phosphorus concentration, N:P ratio), at the subcommunity (chlorophyll a vs. spring overturn P), at the population (algal assay), and at the subcellular (alkaline phosphatase) levels of organization all are consistent with the notion that East and West Twin Lakes were phosphorus limited, at least late in the growing season and should therefore respond to significantly lowered lake concentrations by lowered cell volume and chlorophyll.

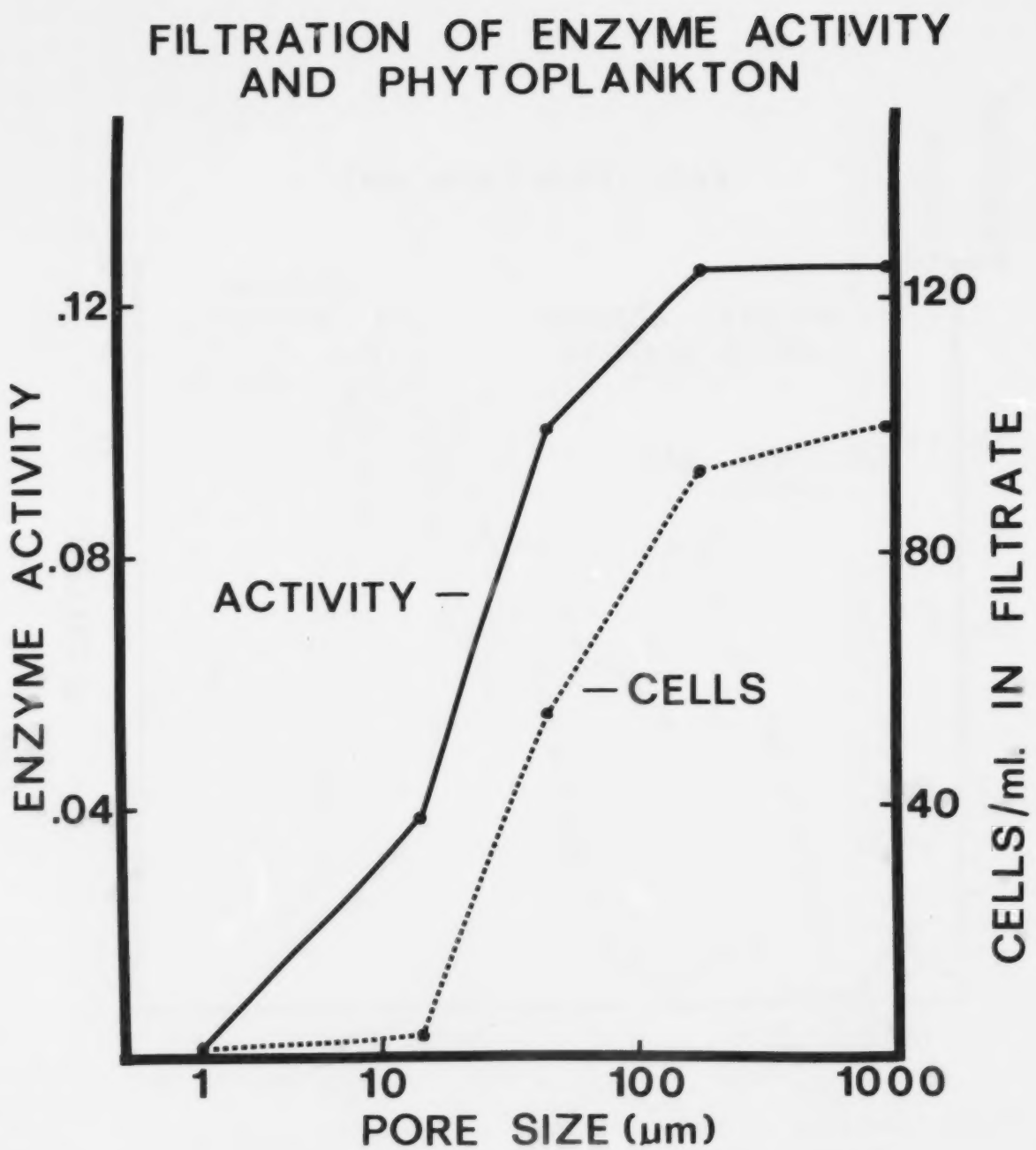

FIGURE 6. Relationship between algal cell size and alkaline phosphatase activity

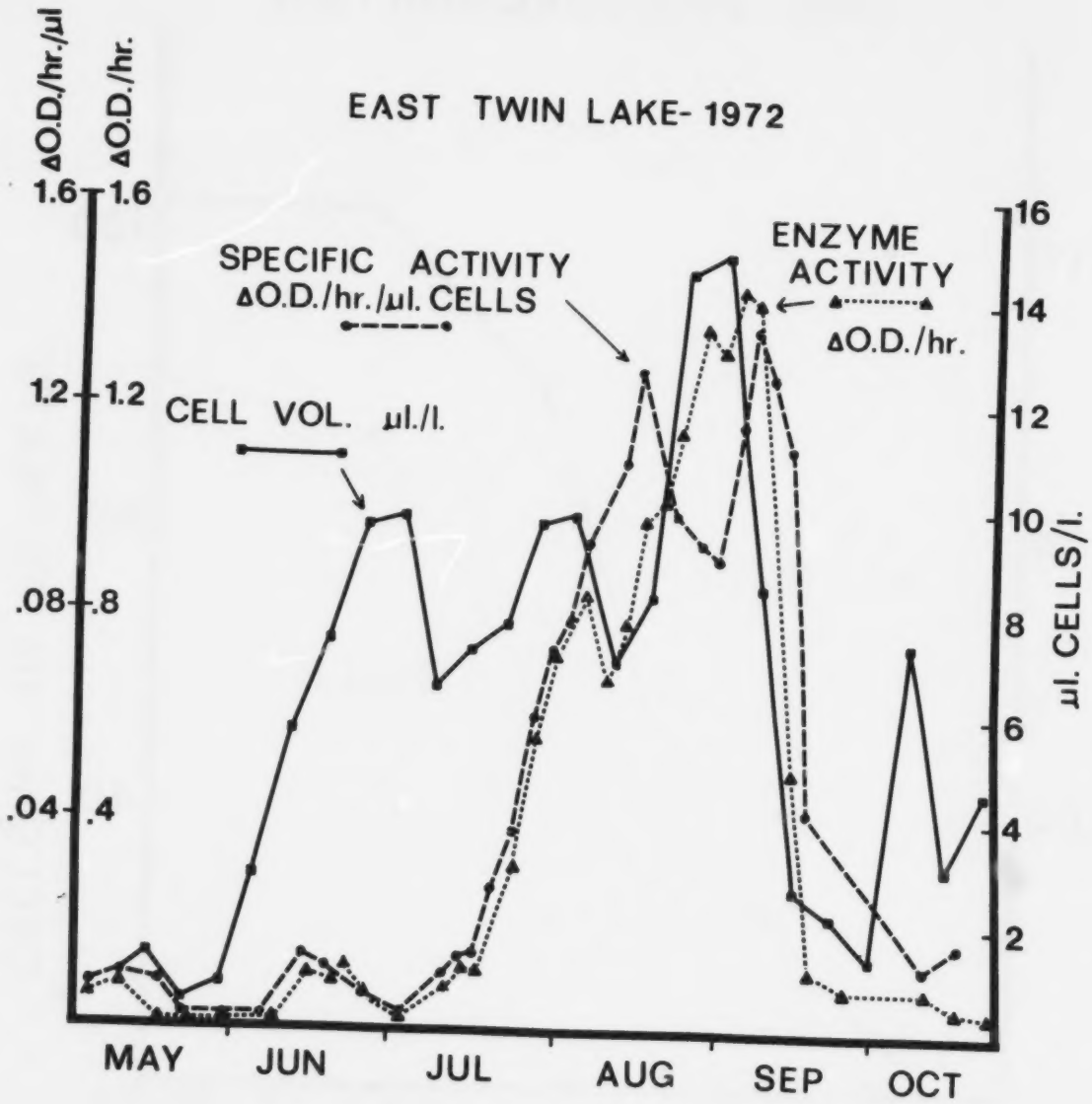

FIGURE 7. Alkaline phosphatase activity, specific activity and cell volume in East Twin Lake, 1972

WATER AND PHOSPHORUS BUDGETS

Septic Tank Diversion

A small sewage treatment plant was constructed outside the watershed to receive the diverted septic tank flows. A few homes were connected to a temporary "package" plant in 1969-71, but the majority of the homes built before sewage service were not connected to the new treatment plant until 1972 (Figure 8). About 85% of the homes were serviced by the end of 1972, and 97% by the end of 1974. All new homes built since sewage service began have been connected to the sewage lines. There are presently a few (10-12) homes, some of them summer cottages, which have not been connected.

Effect of Diversion on Water and Phosphorus Budgets--

Water budget - East and West Twin Lakes are drainage lakes, but their principal sources of water differ. West Twin's water income was about evenly divided between precipitation, groundwater, and streams (Table 8), with groundwater usually the dominant source. East Twin received about half of its annual income from West Twin's outlet, except in 1976 when summer and fall seasons were unusually dry. Nearly all of the stream income to East Twin was from West's outlet. East's overland flow income (Table 9) was much lower than West's because its drainage area was smaller (102 vs 233 ha).

Groundwater, as estimated by the residual in the hydrologic equilibrium, was high for both lakes, particularly East Twin. Because of this large groundwater income plus the discharge of West Twin, the mean water residence time of East was about half that of West (0.58 vs 1.28 years). The areal water load, or mean depth divided by water residence time (Tables 11 and 12), was about twice as high for East than for West (8.85 vs 3.55 meters/year).

The source of the large groundwater income to East Twin was investigated by Szczepanowski (1976). Lake Rockwell, an impoundment 1 km east of East Twin, was filled in 1915 by damming the Cuyahoga River. Otterson (1974) determined that the level of the Twin Lakes rose significantly after this, even though there is a high bluff between Rockwell and East Twin, and no interconnecting surface flows. Szczepanowski installed piezometers in Lake Rockwell, where the two lakes are closest, and estimated a groundwater discharge of 800 M^3 /day towards East Twin.

The seepage from Lake Rockwell may have been partially responsible for maintaining the lake water levels in the Twin Lakes, even after sewage diversion. On other lakes water levels have often dropped following sewage diversion. Water income and water level of neither lake changed in a way which could be attributed to diversion, even though the diversion exported approximately 353 M^3 /day out of the watershed which previously had entered the lakes as overland flow, stream flow, and groundwater.

While the several methods of computing groundwater income to the lakes were in general agreement, the sources of the water were not specifically located. During ice cover unfrozen "windows" were observed in both lakes,

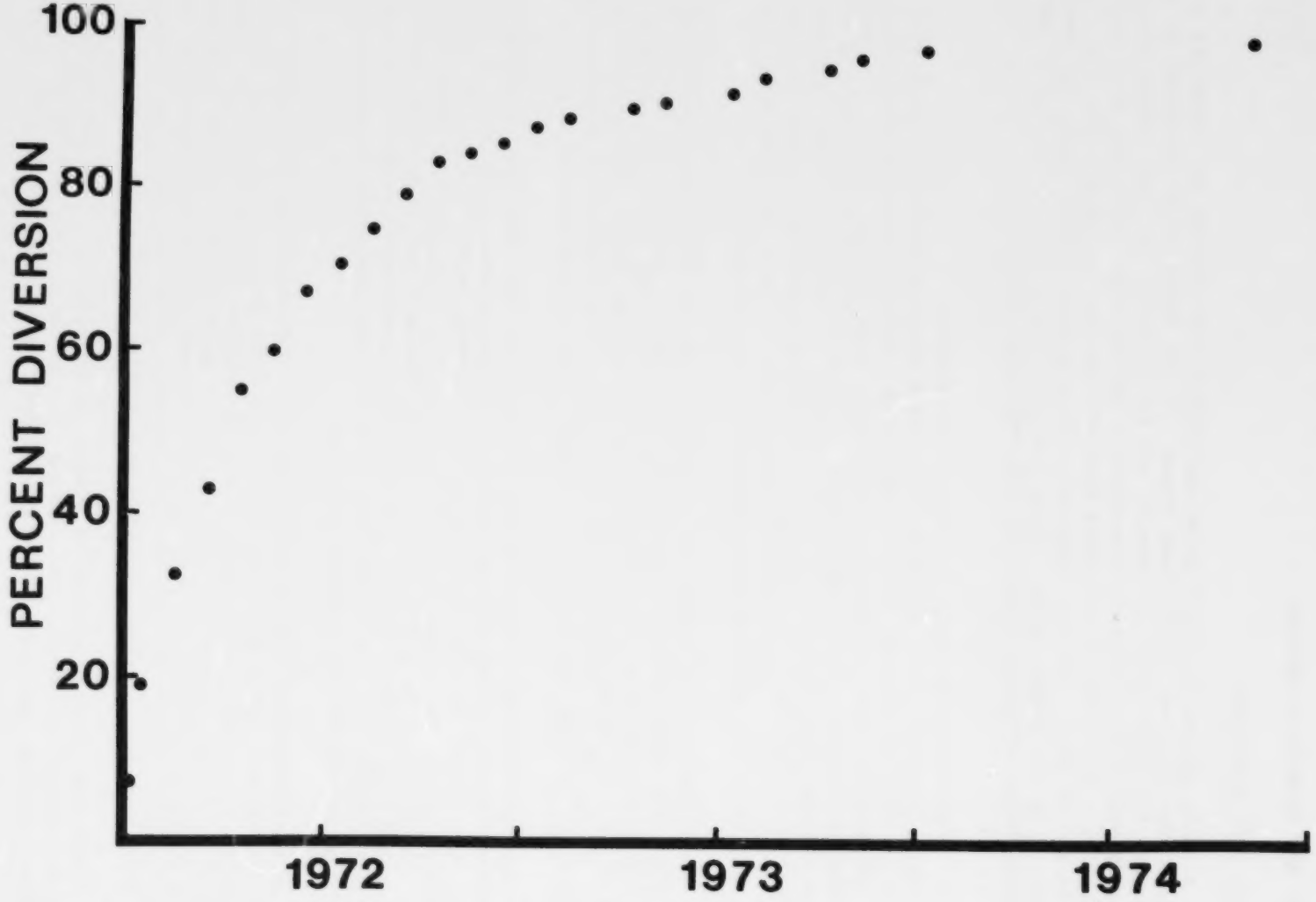

FIGURE 8. Percent septic tank diversion during 1972-1974

Table 8. SOURCES OF ANNUAL WATER ($M^3 \times 10^3$) AND PHOSPHORUS (kg) INCOME OUTGO FOR WEST TWIN LAKE

	1972	1973	1974	1975	1976
1. Annual Water Income	1465.4	1083.0	1679.8	1773.7	1137.3
Groundwater (%)	702.9(48.0)	360.4(33.3)	867.2(51.6)	737.4(41.6)	410.3(36.1)
Streams (%)	361.2(24.6)	392.5(36.2)	336.2(20.0)	464.6(26.2)	209.2(18.4)
Precipitation (%)	325.5(22.2)	317.7(29.3)	395.3(23.5)	399.3(22.5)	370.6(32.6)
Overland (%)	75.8 (5.2)	12.4 (1.1)	81.1 (4.8)	172.4 (9.7)	147.4(13.0)
2. Annual Water Outflow	1182.5	963.1	1544.3	1515.3	935.7
3. Annual Phosphorus Income	142.6	61.7	107.5	155.9	94.3
Groundwater (%)	38.4(27.0)	18.1(29.2)	73.5(68.4)	89.1(57.2)	49.6(52.6)
Streams (%)	86.3(60.5)	35.7(57.9)	19.9(18.5)	32.8(21.0)	19.3(20.4)
Precipitation (%)	6.8 (4.8)	6.8(11.0)	8.8 (8.2)	14.4 (9.2)	8.8 (9.4)
Overland (%)	11.0 (7.7)	1.2 (1.9)	5.2 (4.9)	17.2(11.0)	16.6(17.6)
4. Annual Phosphorus Outflow	129.40	91.26	135.63	100.65	29.84

Table 9. SOURCES OF ANNUAL WATER ($M^3 \times 10^3$) AND PHOSPHORUS (kg) INCOME-OUTGO FOR EAST TWIN LAKE

	1972	1973	1974	1975	1976
1. Annual Water Income	2152.2	2056.9	2971.9	3192.9	2614.4
Groundwater (%)	659.4(30.6)	819.6(39.8)	1050.0(35.3)	1260.8(39.5)	1230.4(47.1)
Streams (%)	1202.7(55.9)	980.9(47.7)	1574.2(53.0)	1541.5(48.3)	1027.1(39.3)
Precipitation (%)	257.2(12.0)	251.1(12.2)	312.4(10.5)	315.5 (9.9)	292.9(11.2)
Overland (%)	33.0 (1.5)	5.4(<1.0)	35.3 (1.2)	75.1 (2.4)	64.2 (2.5)
2. Annual Water Outflow	1978.7	1921.9	2515.1	2996.6	2446.0
3. Annual Phosphorus Income	180.8	126.8	219.4	218.4	127.4
Groundwater (%)	33.0(18.3)	29.2(23.0)	73.2(33.3)	71.8(32.9)	79.9(62.7)
Streams (%)	137.6(76.1)	92.1(72.6)	137.0(62.4)	128.9(59.0)	33.3(26.1)
Precipitation (%)	5.4 (3.0)	5.4 (4.2)	7.0 (3.2)	11.4 (5.2)	7.0 (5.5)
Overland (%)	4.8 (2.6)	0.2 (0.1)	2.3 (1.0)	7.5 (3.4)	7.2 (5.7)
4. Annual Phosphorus Outflow	142.29	108.99	157.44	177.50	135.45

suggesting that fairly high volume springs entered the lakes. These would not be detected by piezometers, so that the rate of seepage from Lake Rockwell, and possibly from the four small lakes above West Twin, could have been much higher than Szczepanowski (1976) estimated. This view further substantiates the decision to assign groundwater income to all unmeasured income, rather than depend upon in-lake and lakeshore piezometer measurements which did not sample higher volume springs.

Phosphorus Budget - Phosphorus entered the Twin Lakes via precipitation, dry fallout, and groundwater and stream flows (Tables 8 and 9). Dry fallout was not monitored.

The annual average concentration of phosphorus in precipitation (Table 10) did not change. The slightly higher concentration in 1975 probably reflects the much larger number of determinations in that year. Overall, the annual aeolian incomes were underestimated since some brief storms after prolonged temperature inversions were not measured, except in 1975, and dry fallout was not measured.

A network of piezometers was established to monitor changes in the phosphorus contributions from groundwater. The mean annual concentration in West Twin's groundwater increased from 1972 to 1975 (Table 10). Changes in 1976 were not monitored and were assumed to be identical to 1975. Mean annual phosphorus concentrations in East Twin's piezometers also increased (Table 10) instead of the expected decrease.

Table 10. MEAN ANNUAL PHOSPHORUS CONCENTRATION IN WATER SOURCES

	µg P/l			
	WEST TWIN			
	Groundwater	Streams	Precipitation	Overland
1972	55	239	21	145
1973	50	91	21	97
1974	85	59	22	64
1975	121	71	36	100
1976	121	92	24	113
	EAST TWIN			
	Groundwater	Streams	Precipitation	Overland
1972	50	114	21	145
1973	36	94	22	37
1974	70	87	22	65
1975	57	84	36	100
1976	65	32	24	112

To determine the impact of nutrient diversion on groundwater quality, especially in the vicinity of septic systems, a series of monitoring wells was drilled above, in, and below a leach field. The monitoring network was set up six weeks prior to connection of the sewers. The boundaries of the leach field were found using electrical earth resistivity techniques. All effluent was diverted from the septic system after sewer hook-up, but the leach field, by request, was not disturbed, and samples of the wells were taken weekly for the first two months and then monthly.

Several important factors about pre-diversion phosphorus income are indicated from this monitoring. First, the water table was encountered in the leach field 0.5 meters below the surface. Six meters above the leach field the water table was about 3 meters below the surface and 3 meters downslope from the leach field, the water table was about 2.5 meters below the surface (Figure 9). These observations indicate a perched water table in the leach field, perhaps caused by an organic mass which reduced the permeability of the leach bed. This situation was common throughout the watershed, as evidenced by soggy lawns even during dry seasons. Nutrient-rich materials were probably washed from the lawns to ditches and streams, and thence to the lakes, accounting for the high concentrations in streams in 1971 and early 1972.

No clay was found in the area of the test leach field; the materials were coarse sand and gravel from surface to 6 meters. Thus septic systems, placed in coarse sand and gravel (soils which are presumably ideal for septic tank-leach field systems) could discharge effluent to the surface as seeps which then contributed to surface water flows, especially during periods of high runoff and overland flow.

The chemical monitoring of the leach field shows the effect of the perched water table and diversion on the groundwater phosphorus concentration. The concentration for the control well 16G, and the well 9 meters below the leach field, 19G, are very close (Figure 10), indicating little phosphorus contamination of the groundwater away from the leach field. Well 17G, situated in the leach field, had a logarithmic drop in concentration immediately after diversion. Well 18G, located at the toe of the field had a slightly higher concentration than the control and dropped little after diversion.

Surface water station 2S (Figure 4) is typical of the urban streams of the watershed where sources include baseflow, stormwater runoff and septic effluent. Stream 2S received the surface seepage from the experimental leach field and run-off from an urban area. Figure 11 illustrates the relationship between reduction in water discharge, phosphorus concentration, and the number of sewer connections from January 1972 through February 1973. In this interval the mean water discharge from 2S dropped from over 400 M^3 /day, to less than 40 M^3 /day. The temporary rise in concentration from June through August 1972 might be due to construction of sewer lines in the area during these months (Buller, 1974).

Figure 12 is a plot of phosphorus output past the weir at station 2S for the same period of time. The output dropped from almost 80 kg P in January

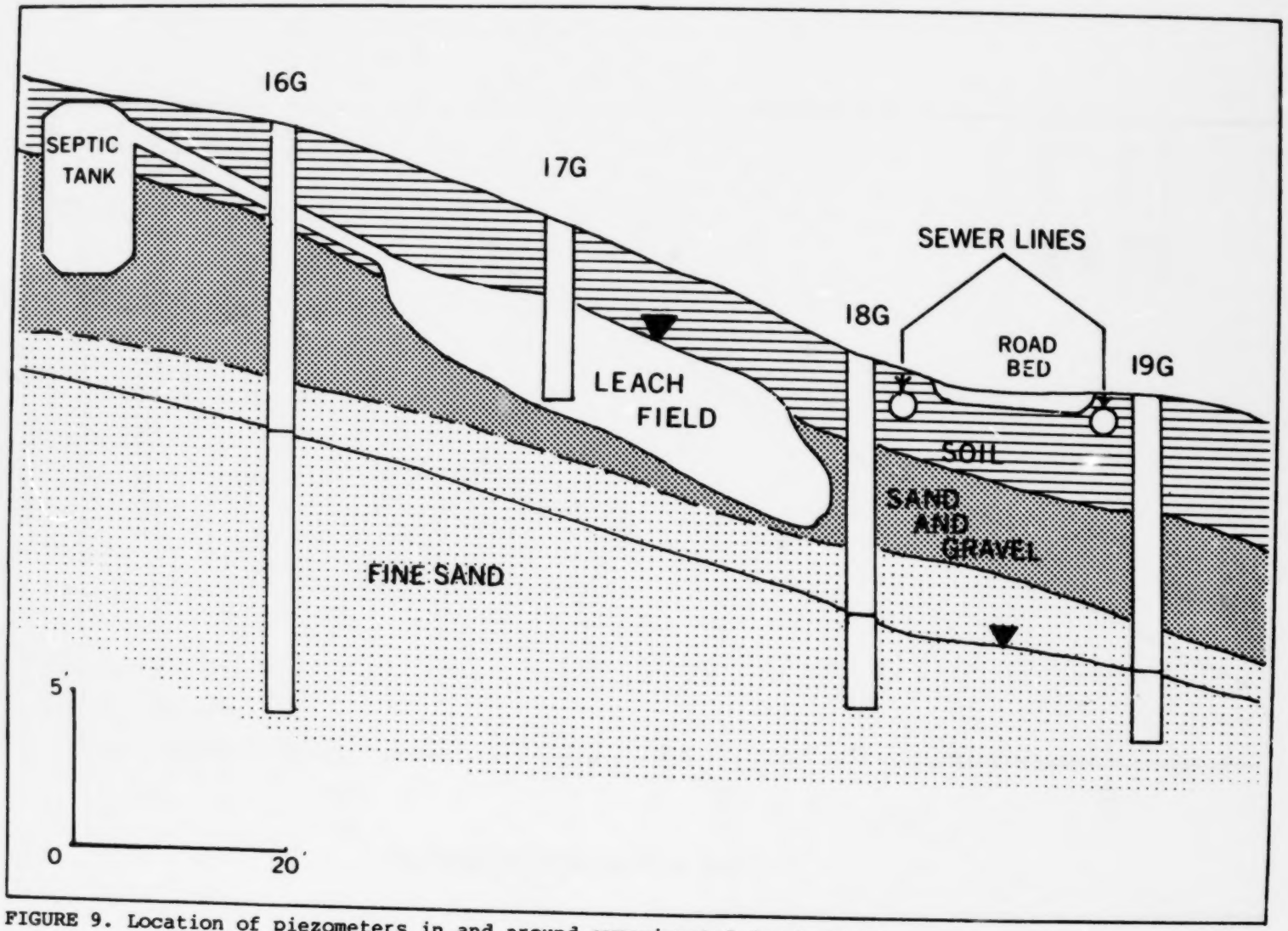

FIGURE 9. Location of piezometers in and around experimental leach field

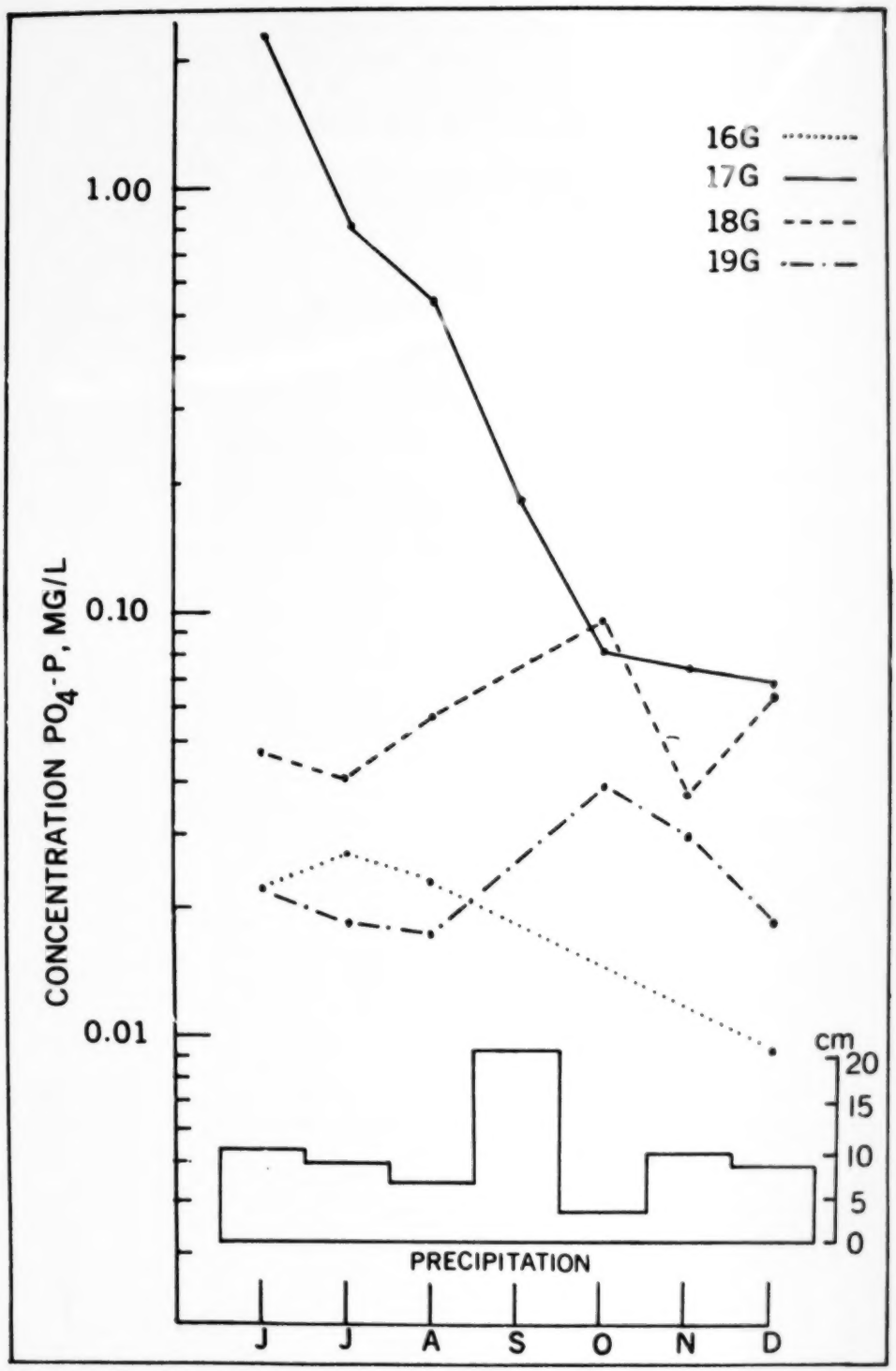

FIGURE 10. Total phosphorus concentration above, in, and below experimental leach field after diversion

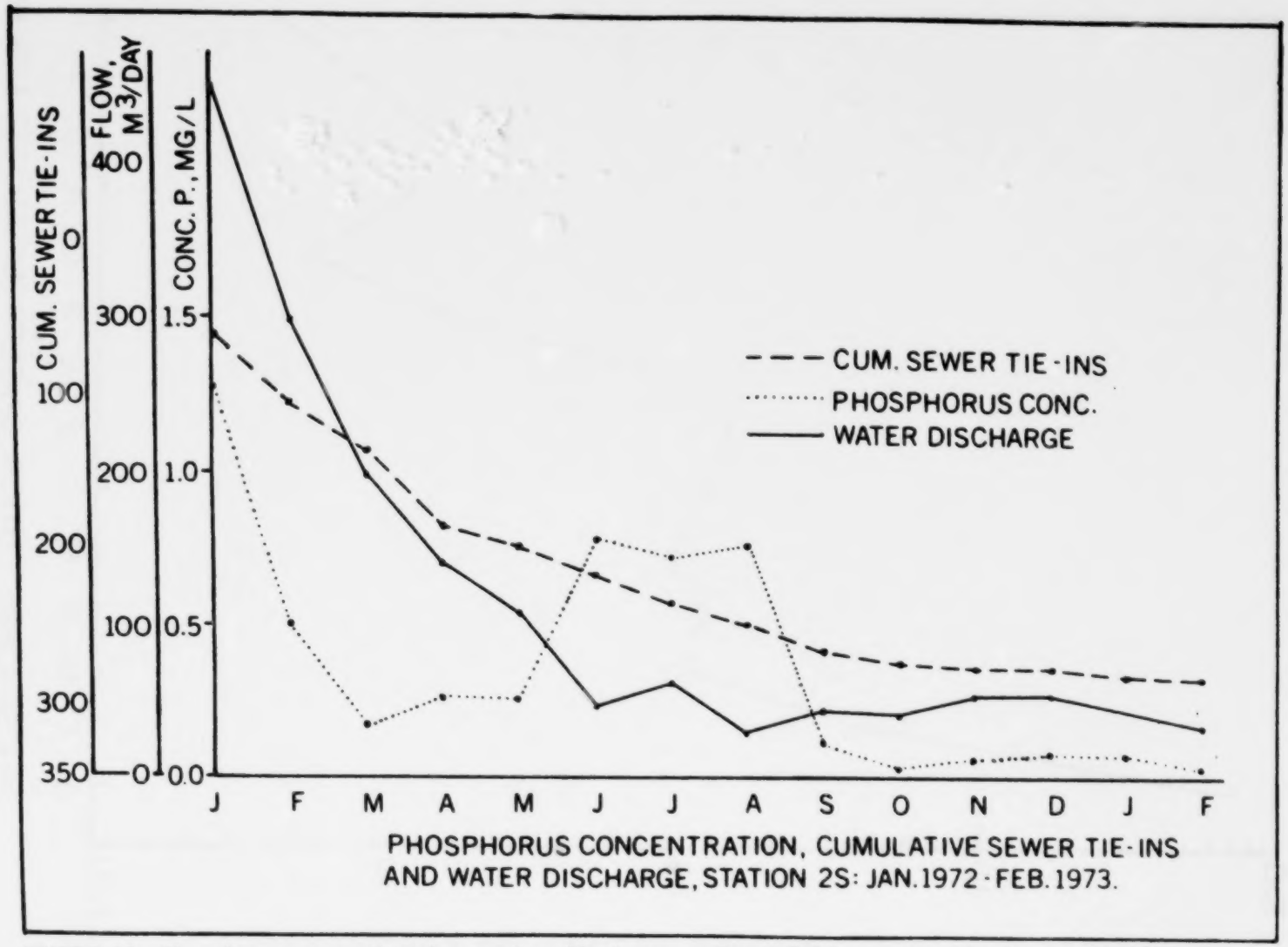

FIGURE 11. Phosphorus concentration, water discharge and cumulative sewer tie-in, stream 2S

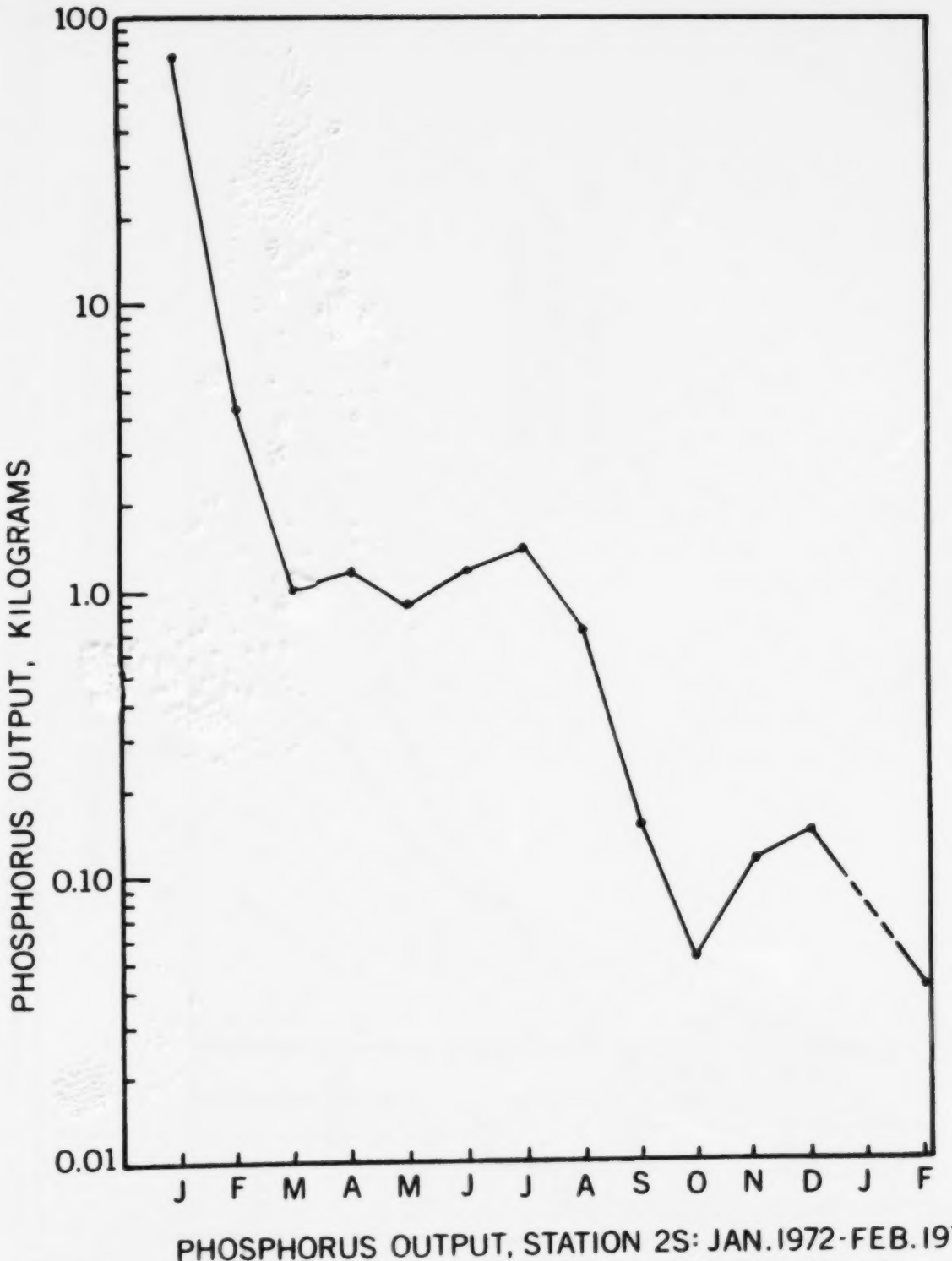

FIGURE 12. Phosphorus discharged by stream 2S to West Twin after diversion

Table 11. WATER - PHOSPHORUS BUDGETS OF WEST TWIN LAKE 1972-1976

	1972	1973	1974	1975	1976
1. Annual Water Inflow $M^3 x10^3$ (Q_i)	1465.4	1083.0	1679.8	1773.7	1137.3
2. Annual Water Outflow $M^3 x10^3$ (Q_o)*	1182.5	963.1	1544.3	1515.3	935.7
3. Water Residence (yrs) $Vol/Q_o = T_w$	1.27	1.56	0.97	0.99	1.60
4. Areal Water Load $\bar{Z}/T_w$ (M/yr)	3.42	2.78	4.47	4.38	2.71
5. Annual Phosphorus Income (kg) (J)	142.6	61.7	107.5	155.9	94.3
6. Annual Phosphorus Outflow (kg)	129.4	91.3	135.6	100.6	29.8
7. Phosphorus Retention $\frac{In-Out}{In}$	0.09	-0.48	-0.26	0.35	0.68
8. Phosphorus Areal Loading (gm $P/M^2/yr$)	0.42	0.18	0.32	0.46	0.28
9. Mean Annual Inflow Concentration (μg P/l) (J/Q_i)	97.3	57.0	64.0	87.9	82.9

*Does not include water lost by evaporation

Table 12. WATER - PHOSPHORUS BUDGETS OF EAST TWIN LAKE 1972-1976

	1972	1973	1974	1975	1976
1. Annual Water Inflow $M^3 x10^3$ (Q_i)	2152.2	2056.9	2971.9	3192.9	2614.4
2. Annual Water Outflow $M^3 x10^3$ (Q_o)*	1978.9	1921.0	2515.1	2996.6	2446.0
3. Water Residence (yrs) $Vol/Q_o = T_w$	0.68	0.70	0.54	0.45	0.55
4. Areal Water Load $\bar{Z}/T_w$ (M/yr)	7.40	7.19	9.31	11.18	9.15
5. Annual Phosphorus Income (kg) (J)	180.8	126.8	219.4	218.4	127.4
6. Annual Phosphorus Outflow (kg)	142.3	109.0	157.4	177.5	135.5
7. Phosphorus Retention $\frac{In-Out}{In}$	0.21	0.14	0.28	0.19	-0.06
8. Phosphorus Areal Loading (gm $P/M^2/yr$)	0.67	0.47	0.82	0.81	0.47
9. Mean Annual Inflow Concentration (μg P/l) (J/Q_i)	84.0	61.7	73.8	68.4	48.7

*Does not include water lost by evaporation

1972 to approximately 0.05 kg P in February 1973. Following the rapid change in total phosphorus in 1972, concentrations in stream 2S remained in the range of 50 to 70 µg P/l. These concentrations represent washoff from streets and lawns. At least one other stream entering West Twin (5S) was also influenced directly by sewage and runoff in a way similar to 2S.

The main source of phosphorus to West Twin was from streams in 1972 and 1973 and groundwater in 1974 through 1976 (Table 7). The mean annual phosphorus concentration in streams in 1972 was very high, particularly the two streams (2S, 5S) draining urban areas rather than those discharging from upland lakes (4S, 3S). Income from streams in 1973-76 was 50-75% lower than in 1972 but mean total annual inflow concentration was as high in 1975 and 1976 as in 1972, due in large part to the increased concentration in groundwater (Table 10). The reasons for this increase are not known.

The mean annual phosphorus concentration in the stream (8S) entering East Twin Lake fell steadily from 1972-1976 (Table 10), with the 1976 concentration 25% of the 1972 concentration, due to the alum treatment of West Twin in 1975. The concentration of phosphorus in other East Twin sources did not decline so that the mean annual inflow concentration, as in West Twin, remained fairly high through 1975.

In summary these drainage lakes received a significant volume of water from seepage and springs. The quantity of phosphorus in groundwater as it enters the lake was not measured, but was estimated from concentration in piezometers on the lake margin. Concentration generally increased over the study period. Surface water flows from urban areas were high in phosphorus before diversion, presumably from upward seepage of effluent from leach fields to the ground surface, where it was carried away as run-off to streams. Stream concentration fell over the study period. Total phosphorus income per year was variable from year to year and average inflow concentration to West Twin was as high in 1976 as in 1972. (Tables 11 and 12).

Vollenweider phosphorus loading model - Vollenweider (1976) has suggested that the critical factor leading to eutrophy in phosphorus-limited lakes is the total phosphorus concentration, a value depending on average annual inflow phosphorus concentration and the water residence time. Eutrophic lakes tend to have high average incoming concentration, moderate residence times, and generally have a phosphorus concentration greater than 20 µg P/l. Vollenweider (1976) designated regions of a phase diagram (mean annual inflowing phosphorus concentration vs water residence time) to show the transition from oligotrophic to eutrophic, divided by boundaries designated as "excessive loading" (those which result in a potential concentration greater than 20 µg P/l) or "permissible loading" (those which result in a potential concentration less than 10 µg P/l).

Using data from Tables 11 and 12, trajectories were plotted with 1972 as the initial year for each trajectory (Figure 13). Over the duration of this study the phosphorus loading to each lake was excessive, and there was no trend from year to year, other than a small decrease from 1972, to suggest that septic tank diversion resulted in a significant decrease towards

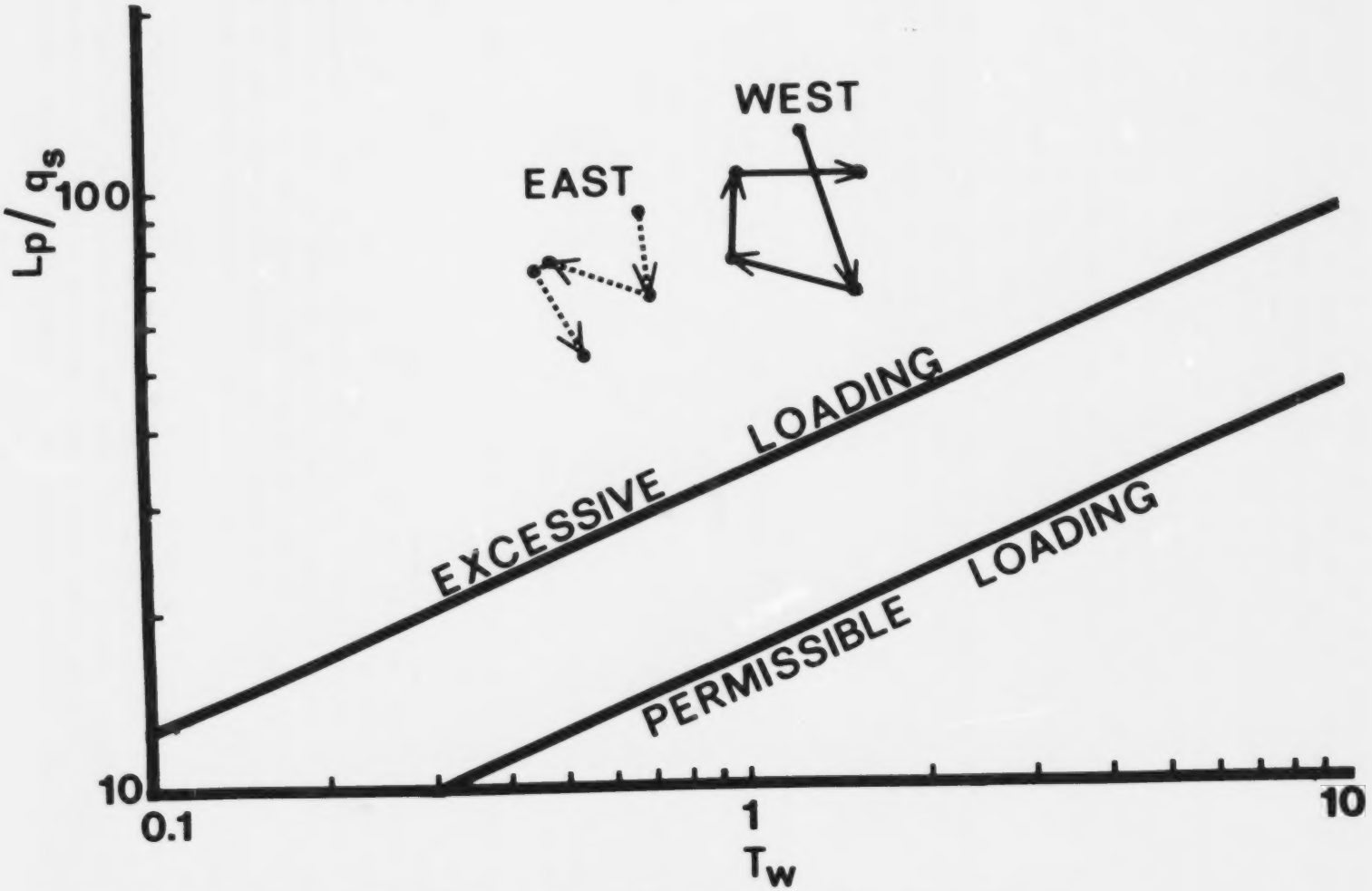

FIGURE 13. Positions of East and West Twin Lakes on Vollenweider (1976) loading diagram, 1972-1976

"permissible" loading. This observation is consistent with the small changes noted in mean annual incoming concentration to both lakes after diversion as well as with the small changes in lake concentration and in biotic indicators of the degree of eutrophication.

EFFECTS OF DIVERSION AND TREATMENT ON PHOSPHORUS

Immediate effects of aluminum sulfate treatment - West Twin Lake was monitored intensely just before, during, and after the aluminum sulfate treatment, and then at weekly intervals through 1976. The immediate responses of the lake were primarily chemical, and differed from any expected longer term response in that there was no immediate effect on biotic indicators of eutrophication in the epilimnion as measured by cell volume, seston, chlorophyll, or transparency.

Aluminum hydroxide particles appeared in deep water samples for several days after treatment. After two weeks the lake was examined through the use of SCUBA. The epilimnion at this time was murky due to an algal bloom, but the hypolimnion, in contrast to the absence of light in pre-treatment dives, was clear. The lake bottom could be seen from about the 5 M level. The mud had a uniform "carpet" of aluminum hydroxide floc about 1 cm thick with no bare spots, and it resembled a moonscape with pock marks where anchors had been dropped before treatment. There was no floc in water less than 5 M deep. Later, the floc consolidated with the sediment so that by 1976 floc particles could only be observed in dredge samples.

Dissolved aluminum remained below detection limits at all depths and locations, with the exception of one sample at 7 M just after application, where a concentration of 2.4 µg Al/l was observed (Table 13).

Table 13. PRE- AND POST TREATMENT ALUMINUM CONCENTRATIONS (µg Al/l) IN WEST TWIN LAKE AFTER 5 METER ALUMINUM SULFATE APPLICATION

	Date (1975)					
DEPTH (M)	7-10	7-29*	7-29**	7-30	7-31	8-1
Surf.	0	0	0	0	0	0
2	0	0	0	0	0	0
4	0	0	0	0	0	0
7	0	0	2.4	0	0	0
10	0	0	0	0	0	0

* 1 hour before treatment
** 1 hour after treatment

The alkalinity and pH of water below 5 M were altered by the alum treatment (Figure 14). Hypolimnetic alkalinity before treatment ranged from 110-140 mg $CaCO_3$ /1. Just after application alkalinity at 6 meters was 60 mg $CaCO_3$ /1, and the hypolimnion below this depth remained at about that alkalinity until fall circulation. Pre-treatment alkalinities returned after circulation and have remained at that level through 1976. The hydrogen ion concentration increased slightly in the hypolimnion, but returned to pre-treatment levels within a month (Figure 14).

Sulfate concentration in the hypolimnion was increased from a pre-treatment volume-weighted average of 35 mg SO_4 /1, to a post-treatment average of 50 mg SO_4 /1. The concentration below 7 M a month after application was over 100 mg SO_4/1, and averaged 50 mg/1 during the fall, in contrast to pre-treatment fall averages of 40 mg SO_4/1 (Table 14). Winter concentrations in earlier years, as well as winter 1975-76, averaged 50 mg SO_4/1. Summer 1976 concentrations were almost twice as high as previous years. There was no change in the sulfate concentration of East Twin.

No toxic effects, such as fish kills, were observed after the application and fishermen report fishing to be as good just after treatment and in 1976 as before treatment.

The concentrations of phosphorus fractions in West Twin Lake were promptly affected by the alum application (Figures 15 and 16). Soluble reactive phosphorus (SRP) in the epilimnion before application ranged from 3-5 µg P/1, and in the hypolimnion, up to 500 µg P/1. Filterable total phosphorus was about 20 µg P/1 in the epilimnion before treatment and above 400 µg P/1 in the hypolimnion (no measurements of FTP fraction were taken at 8 and 9 meters so that SRP, which was measured at all depths appears to be higher). After treatment, the concentration of SRP and filterable total phosphorus in the epilimnion remained unchanged, but were greatly lowered in the hypolimnion. The lowest concentration was reached about one week after application. One year later hypolimnetic concentrations were slightly higher, but were still 10% or less of 1975 pre-treatment levels (Figure 15).

Total phosphorus concentration in the hypolimnion was also greatly lowered after the application (Figure 16) and remained low through 1976 even though oxygen was absent in the hypolimnion.

Effects on Phosphorus Content of the Lakes - The phosphorus content of the Twin Lakes varied in a regular pattern over each year (Figures 17 and 18). During spring circulation phosphorus content steadily fell to an annual low, just prior to the onset of stratification, then increased, most of the increase being in the hypolimnion, to a late summer peak. In some years the increase continued through the fall to a winter peak. The spring decrease began shortly after ice-out.

The daily rate of increase in phosphorus content over the summer interval became less in both lakes each year after septic tank diversion, except in West Twin in 1975 when the interval considered was 40 days shorter than other

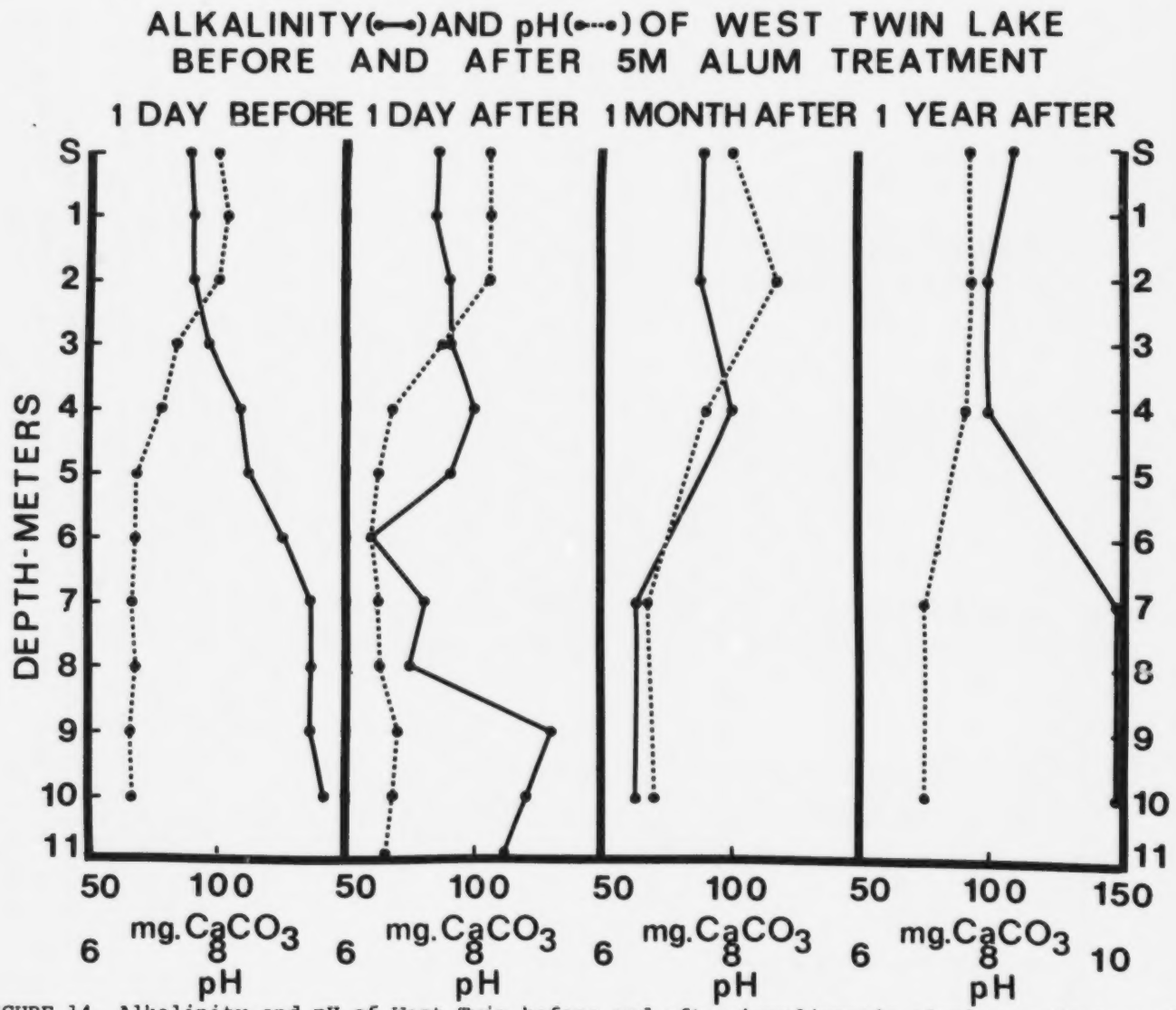

FIGURE 14. Alkalinity and pH of West Twin before and after hypolimnetic aluminum sulfate application

Table 14. SULFATE (mg SO_4/l) IN EAST AND WEST TWIN BEFORE AND AFTER HYPOLIMNETIC ALUM APPLICATION

	West Twin Lake			
	29 Jul 75 0630	29 Jul 75 1830	28 Aug 75	9 Jul 76
Depth				
S	27.50	35.93	40.60	59.35
1	27.35	37.65	41.80	54.41
2	27.20	34.80	43.00	49.97
3	26.85	34.91	43.00	48.80
4	26.50	34.58	43.00	48.14
5	25.60	97.16	65.07	47.71
6	24.70	121.76	87.13	47.27
7	23.80	31.50	109.20	46.84
8	22.97	34.91	108.17	52.55
9	22.13	30.32	107.13	58.25
10	21.30	39.53	106.10	63.96
11	20.47	39.53	105.07	69.67

	East Twin Lake	
	8 Aug 75	13 Aug 76
Depth		
S	39.40	29.89
1	39.55	34.42
2	39.70	38.94
3	39.75	38.42
4	39.80	37.89
5	35.97	35.19
6	32.13	32.49
7	28.30	29.79
8	26.07	30.18
9	23.83	30.57
10	21.60	30.96
11	19.37	31.35

years (Table 15). Had the alum treatment not stopped the 1975 increase, the peak might have been little higher than the previous year and the rate thus equal to or less than 1974. The reduction in this rate of summer increase was particularly pronounced in East Twin in 1975 and 1976 (Figure 18, Table 15).

The effect of the West Twin hypolimnetic alum application on the annual pattern of phosphorus content and the hypolimnetic summer increase is illustrated in Figure 17. The phosphorus content was lowered by 67% of the peak July 1975 pre-treatment content, and remained at that level through 1976, rising somewhat in summer 1976. Autumnal and vernal circulation did not increase either the phosphorus or the aluminum concentration of the lake.

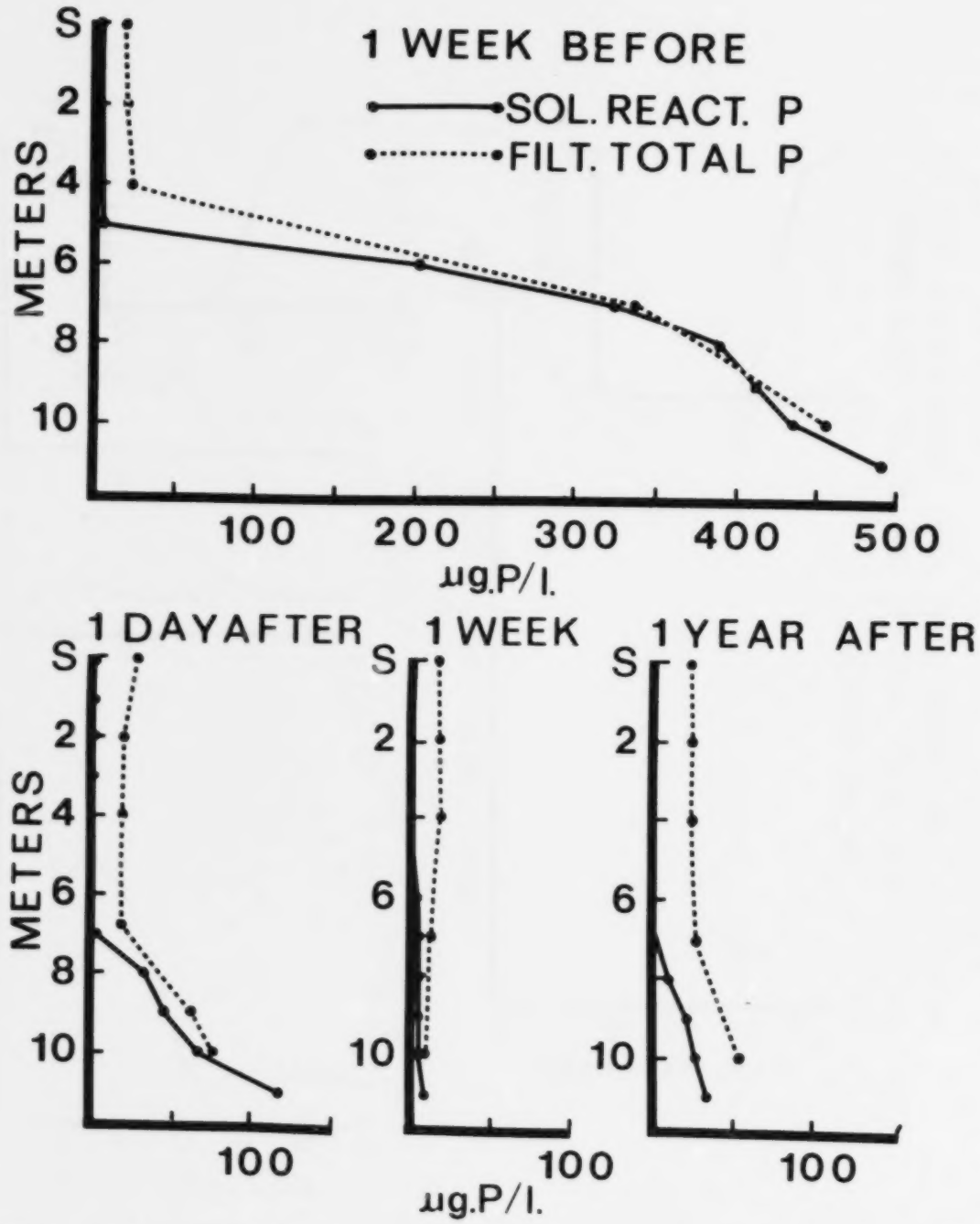

FIGURE 15. Soluble reactive and filterable total phosphorus of West Twin before and after hypolimnetic aluminum sulfate application

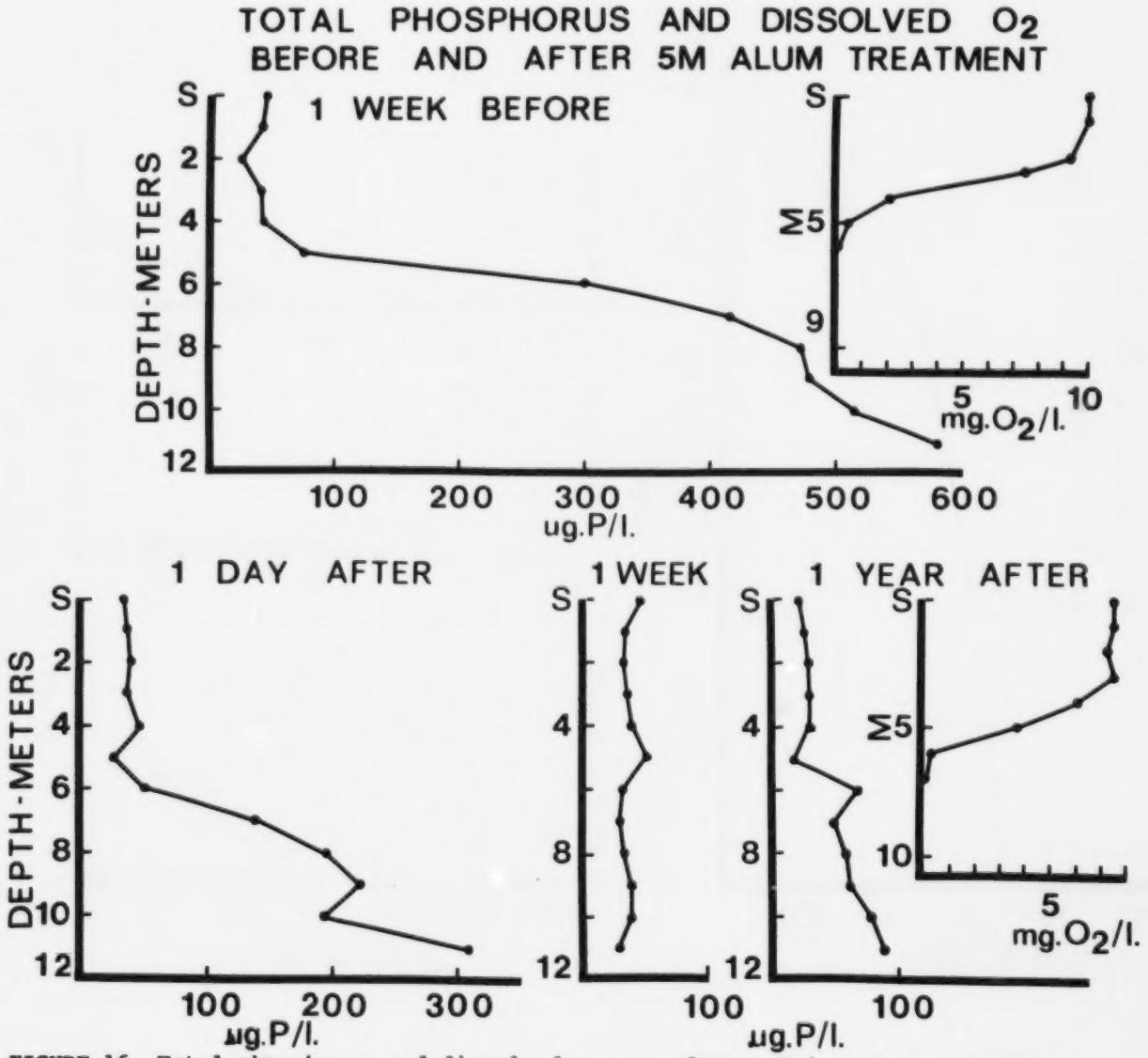

FIGURE 16. Total phosphorus and dissolved oxygen of West Twin before and after hypolimnetic aluminum sulfate application

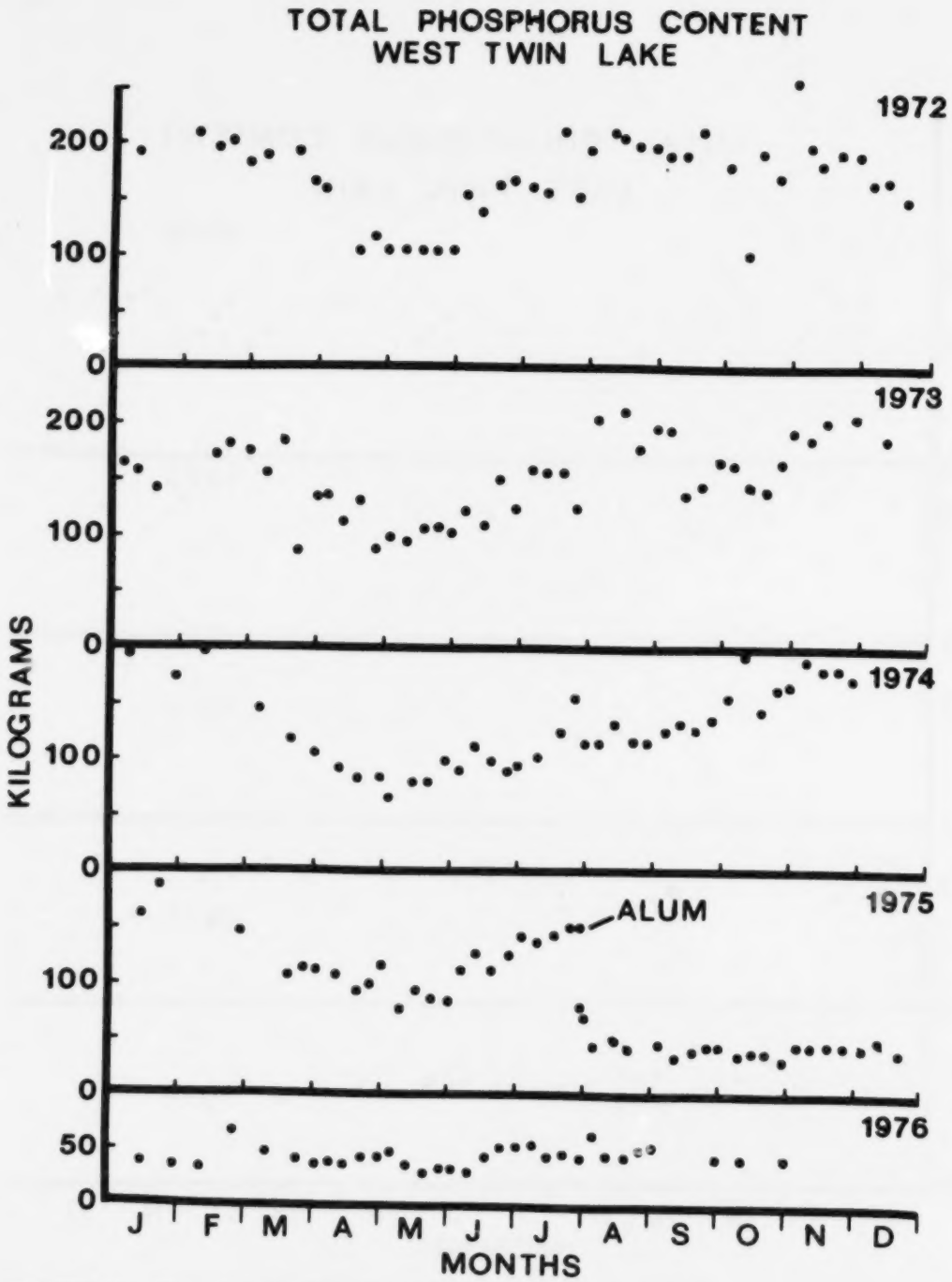

FIGURE 17. Phosphorus content of West Twin Lake, 1972-1976

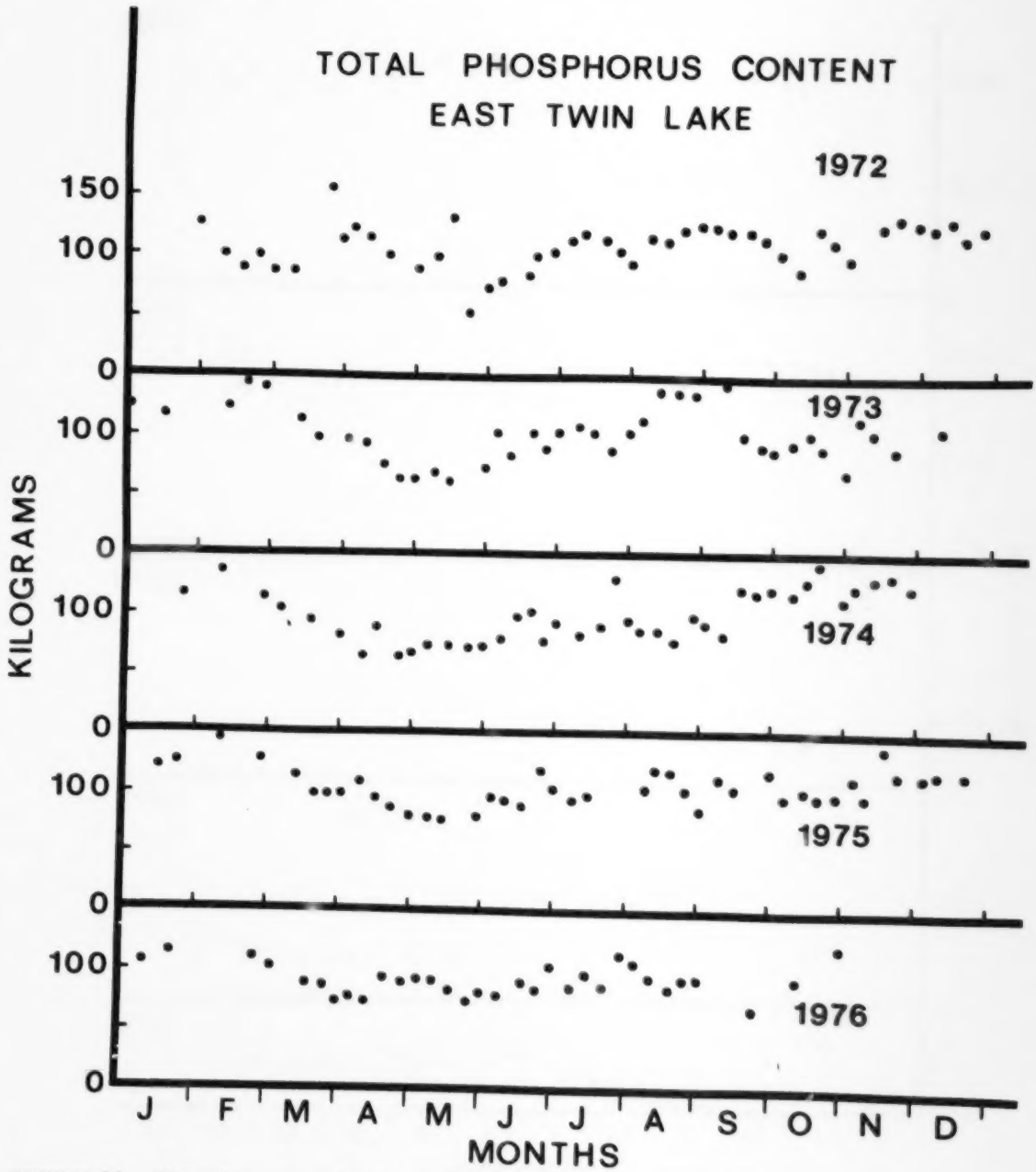

FIGURE 18. Phosphorus content of East Twin Lake, 1972-1976

Table 15. RATE OF INCREASE IN PHOSPHORUS CONTENT SPRING LOW TO SUMMER HIGH gms/day

	West Twin	East Twin
1972	751	769
1973	864	809
1974	430	292
1975	883*	201
1976	228	147

*Up to day of alum application

Changes in the mean volume-weighted seasonal concentration of the three phosphorus fractions during 1971-1976 are illustrated in Figures 19, 20, and 21. Summer means include all lake strata. Changes during 1971-1974 are discussed first, since these changes may be related to diversion.

Mean total phosphorus (Figure 19) in both lakes was high in 1971 before diversion. West Twin, while initially higher in concentration, exhibited a steady decline from 1971 to alum treatment in 1975, while East Twin, after a small drop from 1971 to 1972, remained fairly stable at about 75 µg P/l. Spring mean concentration, which is strongly related to mean epilimnetic chlorophyll (Figure 5), remained at about 70-75 µg P/l in East Twin, while West's declined each year until reaching East's level in 1975.

Mean total phosphorus concentration at spring circulation in both lakes was determined by phosphorus income, lake morphometry, and water budgets. Predicted spring concentration, using the model of Dillon and Rigler (1974b), was always higher than observed, except for 1972 and 1975 in West Twin, and always lower than observed for East Twin (Table 16). However, the correspondence between predicted and observed was close, supporting the conclusion that phosphorus income, as modified by mean depth, flushing, and retention, determined spring concentration. Diversion thus appears to have been responsible for this reduction in the spring mean phosphorus concentration of West Twin.

Seasonal mean filterable total phosphorus (FTP) concentration (Figure 20) was much higher in West Twin in 1973 and 1974, and appeared to decline to East's level by summer 1975. Values for spring and summer 1973 for East Twin were questionable since they tended to exceed SRP and were therefore discarded. Mean seasonal soluble reactive phosphorus (SRP) concentration was almost always higher in West than in East, particularly during the fall (Figure 21). This difference in the fall may have been due to the export of autumn leaves, and leaf decomposition from stream processing and from fires, into West Twin. East Twin, it will be recalled, had no major stream entering it from its subwatershed. Both SRP and FTP concentrations seemed to be relatively unaffected by diversion.

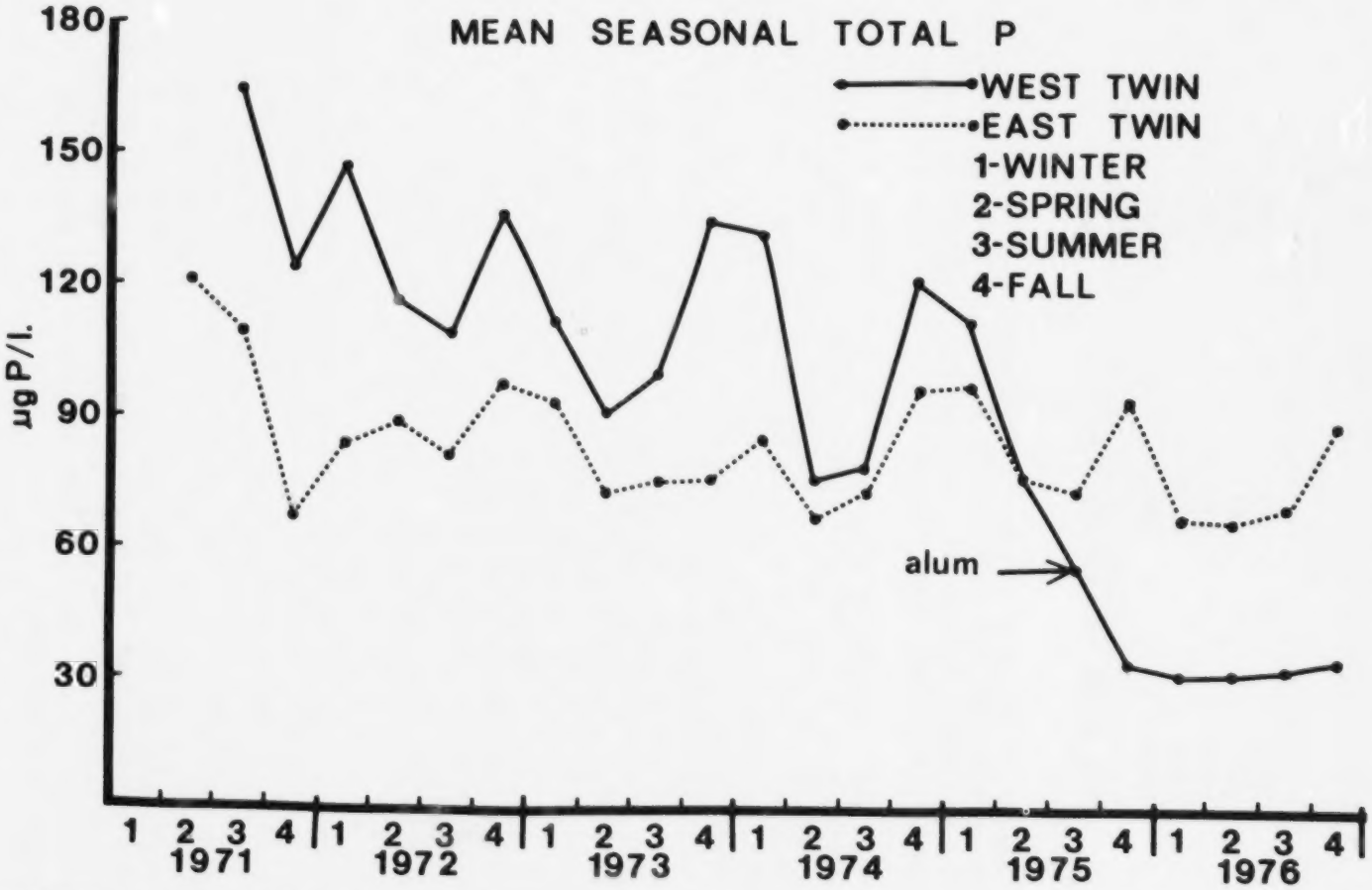

FIGURE 19. Mean seasonal total phosphorus concentration in East and West Twin Lakes, 1971-1976

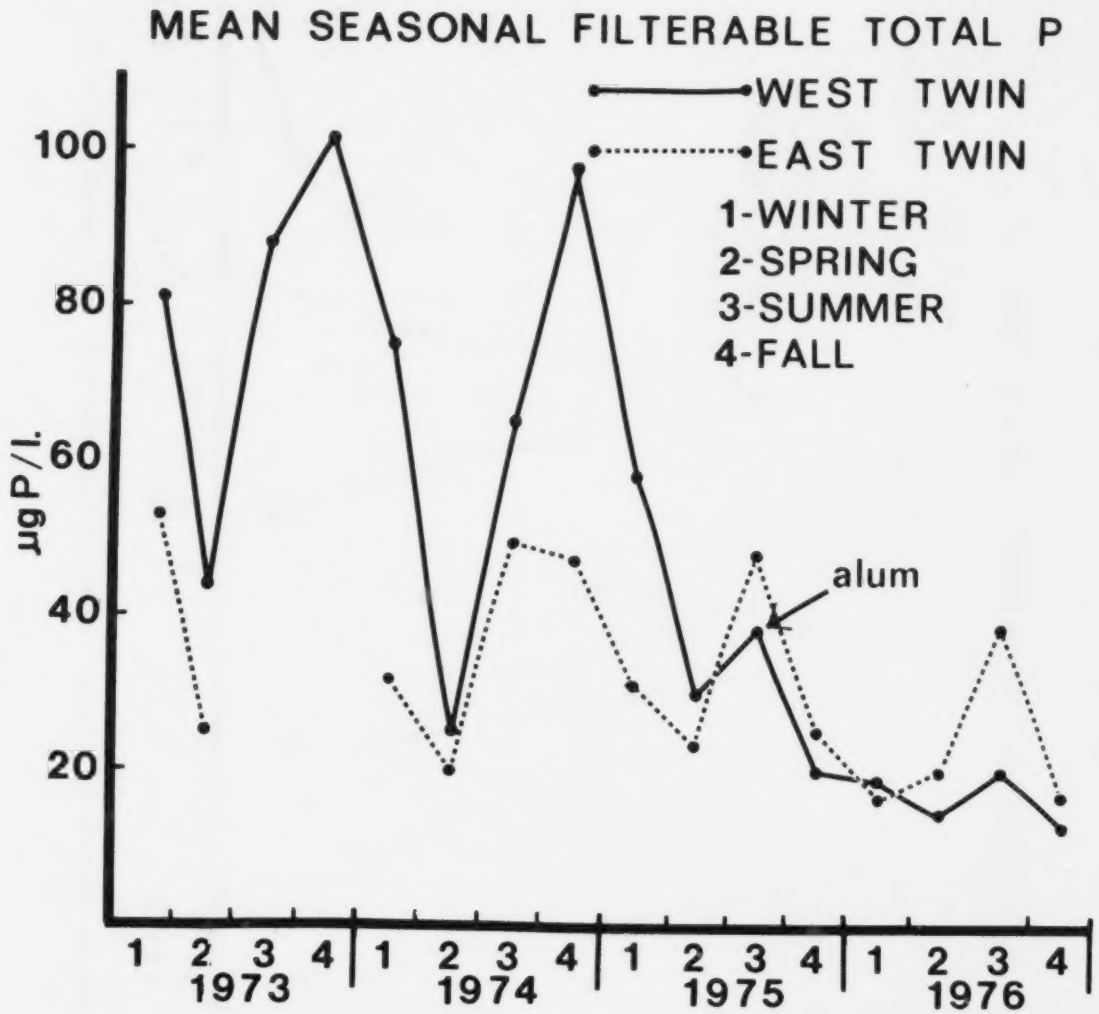

FIGURE 20. Mean filterable total phosphorus concentration in the Twin Lakes, 1973-1976

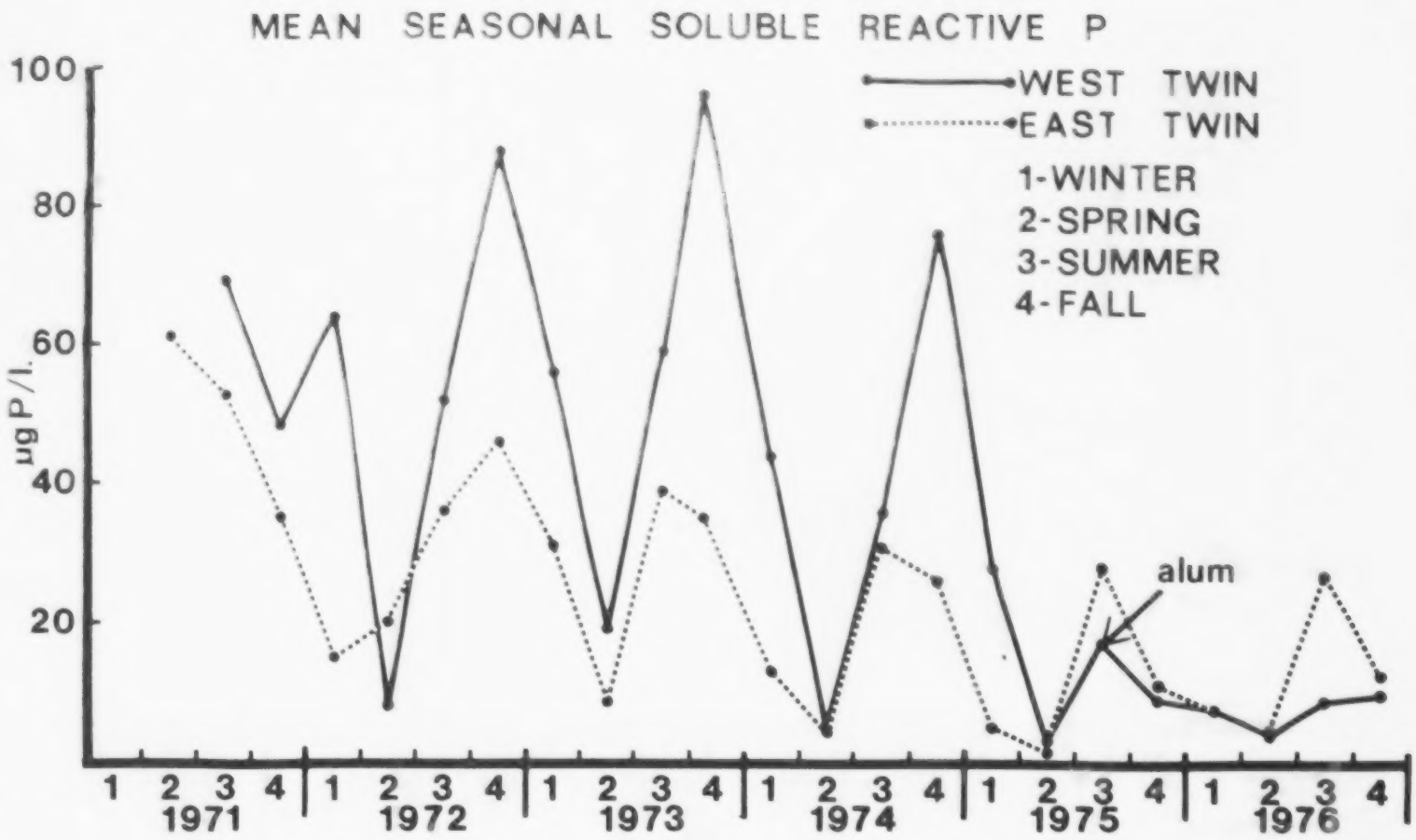

FIGURE 21. Mean soluble reactive phosphorus concentration in East and West Twin Lakes, 1971-1976

Table 16. OBSERVED AND PREDICTED SPRING PHOSPHORUS CONCENTRATIONS IN THE TWIN LAKES (µg P/l)

	WEST TWIN LAKE		EAST TWIN LAKE	
Year	Observed	Predicted	Observed	Predicted
1972	115.6	110.9	88.7	71.5
1973	90.3	96.4	71.9	56.8
1974	74.9	89.2	67.3	54.7
1975	75.8	67.5	75.7	59.1
1976	30.8	32.6	65.4	55.4

From Dillon and Rigler 1974b*

$$*(P) = \frac{L(1 - R_{exp})}{(\bar{Z})\ (1/T_w)}$$

L = P loading in gms $P/M^2/yr$

$$R_{exp} = 1 - \frac{\text{P exported}}{\text{P imported}}$$

$\bar{Z}$ = Mean depth

T_w = Water residence time

A large reduction in the concentration of the phosphorus fractions in the epilimnion should have a significant impact on the symptoms of eutrophication, since the phytoplankton were apparently phosphorus-limited. Average epilimnetic concentrations for each year of monitoring are listed in Table 17. Except for a sharp decline in soluble reactive phosphorus concentration between 1971 and 1972 in East Twin, and a decline in filterable total phosphorus between 1973 and 1974 in West Twin, little change occurred in the epilimnion after diversion until the alum treatment.

The July 1975 application of aluminum sulfate to West Twin's hypolimnion altered the volume-weighted 1976 concentrations of the phosphorus fractions during spring and summer, and the annual average. These concentrations are compared to the last complete pre-treatment year, 1974, for both the treated lake and the downstream untreated lake (Table 18). While both lakes changed from 1974 to 1976, East's in part due to the lowered income from West, West Twin exhibited the larger decrease. All fractions declined, particularly total phosphorus, so that mean concentration of all fractions in West Twin in 1976 was always less than that of East Twin, whereas before treatment it was always greater. The largest decline, apparently due to the alum application, was soluble reactive followed by filterable total and then total phosphorus. The target of the aluminum sulfate treatment, phosphorus in the hypolimnion of

West Twin, was greatly reduced from pre-treatment concentrations and in comparison to the hypolimnion of East Twin.

Table 17. AVERAGE EPILIMNETIC CONCENTRATIONS OF PHOSPHORUS FRACTIONS (µg P/l) (VOLUME-WEIGHTED)

		WEST TWIN LAKE		
		SRP	FTP	TP
1971		13.34	N.D.	40.59
1972		11.39	N.D.	58.09
1973		15.99	40.62	52.73
1974		10.25	29.31	47.46
1975	I*	3.50	26.42	43.80
	II*	5.40	15.55	28.93
	All	4.50	20.72	36.01
1976		8.05	19.16	28.26
		EAST TWIN LAKE		
		SRP	FTP	TP
1971		21.35	N.D.	30.97
1972		8.84	N.D.	44.50
1973		7.78	N.D.	37.61
1974		8.60	24.89	44.44
1975		5.02	23.15	39.38
1976		8.16	19.74	37.97

I = Days from stratification until 29 July treatment
II = Days from treatment to autumn circulation
SRP = Soluble reactive phosphorus
FTP = Filterable total phosphorus
TP = Total phosphorus (See Limnology Methods, p. 26)

However, while mean total phosphorus in the epilimnion of West Twin was 25.5% less in 1976 than East Twin (28.3 vs 38.0 µg P/l), filterable total and soluble reactive phosphorus of the epilimnion were only 2.5% (19.2 vs 19.7 µg P/l) and 1.2% (8.1 vs 8.2 µg P/l) less, respectively (Table 18). Mean annual and epilimnetic concentrations of all fractions after treatment remained within the range expected of eutrophic lakes.

The small change in epilimnetic filterable total phosphorus may be of some significance since low molecular weight organic-phosphorus molecules

found as part of this fraction in the Twin Lakes (Francko and Heath, in press) may serve as sources of phosphorus to some phosphorus-limited blue-green algae (Heath and Cooke, 1975).

In summary, septic tank diversion apparently produced a steady decline in mean total phosphorus of West Twin, and the spring concentration in the lake declined to the level of East Twin's spring phosphorus. However, the 1975 spring concentration in both lakes, before the alum application, was about 75 µg P/l, a level which would produce a mean summer epilimnetic chlorophyll concentration of a moderately eutrophic lake. Thus diversion, while effective in lowering the concentration of phosphorus, did not bring it down over a four year post-diversion period to levels which would result in marked improvement in symptoms of eutrophication.

Internal Phosphorus Loading - The release of phosphorus from anoxic lake sediments has been commonly observed in both laboratory and field experiments and is believed to be a significant internal source of phosphorus. The anoxic sediments of the hypolimnion of West Twin Lake were therefore treated with a maximum dose (as earlier defined) of aluminum sulfate in July 1975, with control of this internal phosphorus loading to the lake as the objective.

The rate of net internal phosphorus loading was obtained by difference from this mass balance equation (Cooke et al. 1977):

$$\Delta P_{lake} = (P_i - P_o) \pm P_{int}$$

where:

P_{lake} = change in the phosphorus content of the lake from the spring low to the summer high.

P_i = phosphorus income

P_o = phosphorus losses via outflow

P_{int} = phosphorus gained (net release) or lost (net sedimentation) due to internal loading.

The rates of net external and internal phosphorus loads, based upon lake surface area, and from the period of spring minimum lake phosphorus content to the summer maximum, are listed in Table 19. Net internal loading (as compared to external loading in Table 19) accounted for 52-105% of the increase in phosphorus content of the lakes during summer. No rates were calculated for 1975 since West Twin was treated at that time and because of the apparently anomalous values for East Twin, which were related to a large loss to groundwater in June. This was the only year when water was lost to groundwater during the summer.

Net summer internal loading rates declined from year to year, with the exception of a small increase in East Twin's 1976 rate, and summer external loading increased. The sum of rates indicates that in 1972 and 1973 phosphorus loading was about twice that of 1974 and 1976.

Table 18. PRE- and POST- ALUM TREATMENT VOLUME WEIGHTED MEAN CONCENTRATIONS OF PHOSPHORUS FRACTIONS (μg P/l)

	WEST TWIN LAKE (treated)				EAST TWIN LAKE			
	Spring	Epi	Hypo	Year	Spring	Epi	Hypo	Year
	1. Total Phosphorus							
Pre-Treatment (1974)	74.9	47.5	420.5	96.6	67.3	44.4	405.6	76.9
Post-Treatment (1976)	30.8	28.3	75.2	32.1	65.4	38.0	268.8	70.2
Percent Change	-58.9	-40.4	-82.1	-66.8	-2.8	-14.4	-33.7	-8.7
	2. Filterable Total Phosphorus							
Pre-Treatment (1974)	25.9	29.3	358.0	69.5	20.0	24.9	N.D.	42.0
Post-Treatment (1976)	14.5	19.2	47.9	18.1	20.5	19.7	177.7	29.7
Percent Change	-43.3	-34.5	-86.6	-74.0	+2.5	-20.9	-45.8	-29.3
	3. Soluble Reactive Phosphorus							
Pre-Treatment (1974)	5.8	10.3	355.7	41.8	5.2	8.6	345.5	24.0
Post-Treatment (1976)	4.0	8.1	42.7	8.4	4.4	8.2	209.8	18.8
Percent Change	-31.0	-21.4	-88.0	-79.9	-15.4	-4.7	-39.3	-21.7

Table 19. NET SUMMER EXTERNAL, INTERNAL AND TOTAL PHOSPHORUS LOADING TO THE TWIN LAKES (mg P/M^2/day)

Year	External	WEST TWIN Internal	Total	Days
1972	0.200	2.007	2.207	126
1973	-0.127	2.668	2.541	126
1974	0.435	0.831	1.266	116
1975		NO CALCULATION - ALUM APPLICATION		
1976	0.637	0.689	1.326	77

Year	External	EAST TWIN Internal	Total	Days
1972	-0.054	2.917	2.863	98
1973	0.207	2.804	3.011	112
1974	0.336	0.750	1.086	125
1975		NO CALCULATION - ALUM APPLICATION		
1976	0.663	1.022	1.685	72

The 1975 alum application to the hypolimnion did not result in the expected sharp decline in 1976 net internal phosphorus loading of West Twin. Over 50% of the summer increase in 1976 was due to internal loading and the rate was only slightly less than the 1974 rate (.831 in 1974 vs .689 mg P/M^2 /day in 1976).

In situ experiments with polyethylene columns, where aluminum sulfate applied to the hypolimnion controlled phosphorus release for 104 days, and the inverted barrel experiments in Dollar Lake where alum treated (in 1974) lake muds exhibited very small phosphorus release (through 1976) when compared to untreated muds, strongly support the conclusion that the release of phosphorus from alum-treated hypolimnetic muds in West Twin was controlled (see Appendix and Kennedy, 1978, for further details of these experiments). However, the similarity in mean epilimnetic total phosphorus concentration between the lakes in 1976 (WTL = 28.3 µg P/l, ETL = 38 µg P/l) (Table 17), and the very low hypolimnetic concentration in West Twin in 1976 (Figure 17 and Table 18) support the conclusion that vertical transport, described by Stauffer and Lee (1973) as a source of internal loading to the epilimnion, was not of great significance in the Twin Lakes. Significant portions of the net internal phosphorus release during the summer must have been from sources other than the anaerobic hypolimnion, since net internal loading continued despite an

effective alum treatment and the concentration of phosphorus in the epilimnion of the two lakes was similar despite dramatic differences in hypolimnetic concentration.

Further studies are needed to confirm this conclusion, but it does suggest an explanation for other observations. For example, it is not intuitively clear why the late winter - early spring phosphorus concentration should be strongly related to mean epilimnetic summer chlorophyll (Dillon and Rigler, 1974a;Figure 5), when that phosphorus is lost from the water column during April and May each year (Figures 17 and 18). However, if the precipitated phosphorus is released as summer internal phosphorus loading, then the strong correlation may be explained. In smaller lakes, groundwater and the biota of the littoral may be especially important in returning phosphorus to the water. Macrophytes have been shown to transport phosphorus from roots through leaves to the water (e.g., McRoy et al. 1972), and fish may release significant amounts (Lamarra, 1975). These studies and others strongly suggest that both abiotic and biotic sources in the littoral zone may be the major contributors of internal phosphorus loading in lakes like the Twin Lakes.

The annual phosphorus retention coefficient (R_p) is related to internal loading. Kirchner and Dillon (1975) developed an equation from the data for 15 Canadian lakes which estimated R_p from the areal water load ($r = 0.94$). In Table 20 the R_p's predicted from their equation and the observed R_p's for Twin Lakes are listed (see also Tables 11 and 12). Both of the Twin Lakes had R_p's far lower than predicted, which means that either incoming phosphorus was not sedimented in great quantities or was re-released after sedimentation as internal loading, and then lost through the outflow. The evidence of internal loading presented here for the summer period supports the belief that phosphorus was rapidly recycled in the Twin Lakes. Further, since it appears that the littoral zone may be a major source of the internal loading, a large amount of sedimentation must also occur here.

Table 20. PREDICTED AND OBSERVED PHOSPHORUS SEDIMENTATION COEFFICIENTS IN THE TWIN LAKES

Year	EAST TWIN		WEST TWIN	
	Observed	Predicted	Observed	Predicted
1972	0.213	0.592	0.093	0.724
1973	0.140	0.597	-0.480	0.760
1974	0.283	0.542	-0.261	0.688
1975	0.187	0.537	0.355	0.681
1976	-0.064	0.562	0.684	0.764

Lakes with a prolonged history of high phosphorus income may become "saturated" with regard to their ability to efficiently retain phosphorus in the sediments and/or in biomass. After the alum treatment the 1976 R_p of West Twin was nearer that predicted by Kirchner and Dillon (1975), supporting a conclusion that the low hypolimnetic concentrations after treatment were due to the retention of material sedimented from the epilimnion, as well as to retention of phosphorus already in the sediments.

Is a hypolimnetic application of aluminum sulfate a useful technique if the hypolimnion, at least in small lakes where large seiches do not occur, is not the only major internal source of phopshorus to the epilimnion? The results of Kennedy (1978) show that phosphorus release from alum-treated anaerobic muds is controlled (see also Appendix of this report). Further, phosphorus which enters the hypolimnion of West Twin from the epilimnion is apparently retained in the mud, since the concentration in the oxygen-free hypolimnion remained low over the summer. As long as the aluminum hydroxide "blanket" is effective in trapping and retaining phosphorus, lake concentration at spring circulation may continue to decrease and reduction of summer algal blooms may result. Longer term observation of the Twin Lakes are needed to establish the effectiveness of the treatment.

The long-term efficacy of the aluminum sulfate method will depend largely on the source of internal phosphorus loading. In large lakes such as Mendota, where the hypolimnion is a more significant epilimnetic source through thermocline erosion (Stauffer and Lee, 1973), hypolimnetic application might be very successful in lowering epilimnetic phosphorus concentration, although very expensive (see Appendix for costs). In small lakes with a larger littoral area relative to total lake area, internal phosphorus loading may be more difficult to control. If the major littoral phosphorus source is groundwater flow through phosphorus-rich sediments, then an application of alum could be very effective. This is a distinct possibility as a source since Lee (1972) has presented evidence which suggests that groundwater income declines exponentially from the shore. However, if the primary sources are biotic, as some current evidence indicates, then phosphorus inactivation with alum may not be sufficient to sharply reduce internal loading.

The littoral area alone of a small lake similar to one of the Twin Lakes, where an accurate nutrient budget is available, should be treated. This would allow some assessment of the significance of abiotic littoral sources of phosphorus to net internal loading.

Effect on Conductance, pH, Alkalinity, and Aluminum

The immediate effect of the alum application on these aspects of water chemistry was described earlier. There appeared to be no longer term effects. Levels of conductance, pH, alkalinity, and dissolved aluminum were found to be at pre-treatment levels in 1976. After the treatment insufficient observations were made on other aspects of water chemistry such as nitrogen forms, except for sulfate. Sulfate concentration in West Twin in 1976 was about twice that of pre-treatment measurements (Table 14).

EFFECTS OF DIVERSION AND TREATMENT ON BIOLOGICAL INDICATORS

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Coliform Bacteria

The streams and the lakes were heavily contaminated with coliform bacteria in the late 1960's and early 1970's before diversion. Presumably these bacteria came from the seepage of septic tanks to the ground surface, in a way similar to that described earlier for phosphorus. In 1969-70 the lakes were closed to contact recreation when colony forming units (CFU) exceeded 200/100 ml.

The fecal coliform content of the peripheral lake wells in 1972 was uniformly high in March and April (up to 80,000 CFU/100 ml along East Twin's east shore to 7800 CFU/ ml along West Twin's west shore). During the summer of 1972 as diversion progressed and as the water table fell due to lowered precipitation, colony counts declined. The change in the concentration of fecal coliform bacteria in three wells, labelled 1G (1.5 meters), 2G (3 meters), and 3G (6 meters) and located along the stream (1S) entering West Twin's lagoon on the west shore (Figure 4), illustrates the effect of septic tank diversion on ground water quality. 1G was always free of fecal coliforms in 1972 and 1973. 2G and 3G, which were deep enough to always be in the water table, had high concentrations in March and April 1972 (6-8000 CFU/100 ml). By August 1972 no CFU could be detected. In July 1973 the highest count was only 450 CFU/100 ml.

The coliform content of stream 2S, which drained a heavily developed housing area on the south shore of West Twin (Table 21) was representative of the other urban streams of the watershed. Colony forming units were very high through the summer of 1972, declined in the fall of 1972, and by 1973 were very low. This paralleled the reduction in phosphorus concentration of this stream. Coliform forming units in lake outflows were usually low (500 CFU/100 ml) in 1972 and very low in 1973 (100 CFU/100 ml).

The lakes had a prompt reduction in fecal coliforms after septic tank diversion. A concentration of 200 CFU/100 ml was last observed in April 1972 at 55% septic tank diversion. After that date counts in surface (0.1 M) water never exceeded 100 CFU/100 ml, and were usually less than 10 CFU/100 ml.

Cell Volume, Chlorophyll, Phytoplankton, Seston, and Transparency

Nutrient diversion, plus any restorative techniques, must bring about a significant reduction in the symptoms of eutrophication during the spring and summer months in order to be considered successful. To assess the impact of septic tank diversion and alum application on the Twin Lakes, indicators related to plankton biomass, (cell volume, chlorophyll, species composition, seston, and Secchi disc transparency) were monitored at weekly intervals during all months from spring 1971 through summer 1976. In this report data for surface waters in the summer (May-November) only will be considered.

Table 21. FECAL COLIFORM COLONY FORMING UNITS (#/100ml) IN URBAN STREAM 2S

1972		1973	
January	NS	January	10
February	48000	February	<1
	14000		<1
March	NS	March	NS
April	8100	April	NS
	8400		
May	5800	May	NS
	5880		
	260		
June	3390	June	<1
	1920		
July	TNTC	July	<1
	2490		<1
	24860		200
	43700		580

NS = No Sample
TNTC = Too Numerous to Count

The Twin Lakes Association treated portions of the lake each year for macrophyte and algae control, by adding endothal and diquat, or endothal and silvex, to the beach area near stream 2S, to the lagoons, and to two docks on the north side of West Twin. In East Twin the same materials were added to the area around the inflow of West Twin and to the lake outflow area. The residents objected to stopping the treatments for the purposes of this investigation. Additionally, some lake shore residents added herbicides. One resident at East Twin added 200 pounds of granular copper sulfate over one summer to his beach area. At no time during this study was an herbicidal chemical added to either lake in the vicinity of the deep open water sampling station. While most shore treatments were added when the lakes were calm, some of the herbicides must have drifted to the sampling area. In particular, summer 1973 was a year of extensive treatment and values for cell volume and chlorophyll reflect this.

Surface cell volume at the beginning of the study was very high in West Twin (Figure 22). By summer 1972, when about 70% of the septic tank diversion was completed, cell volume in West Twin had fallen to less than half that of the previous summer, and to about the same level as East Twin. Cell volume in 1973 was very low and may have been due to frequent and heavy algicide application to control dense mats of Rhizoclonium along the shore. Cell

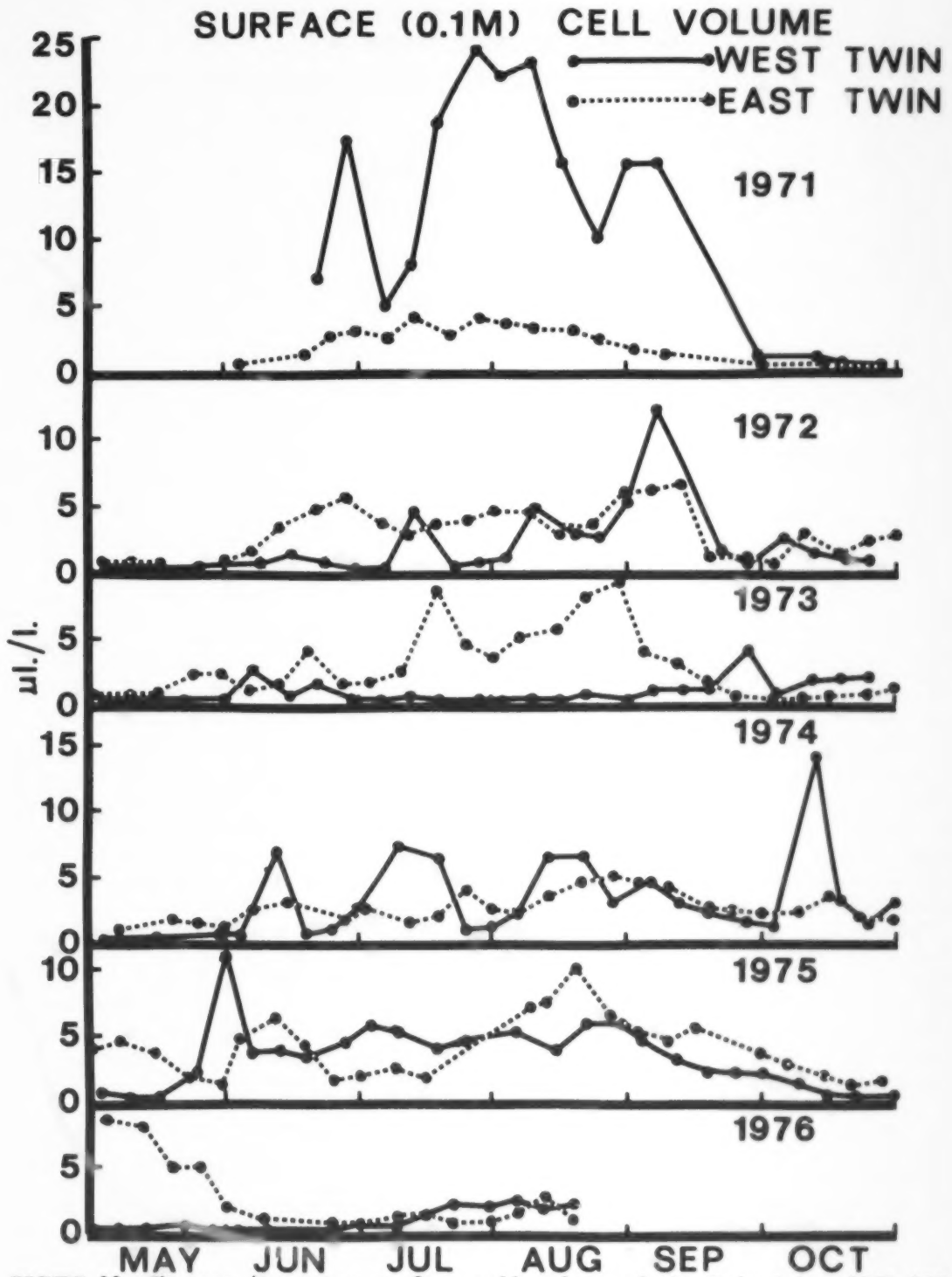

FIGURE 22. Changes in summer surface cell volume of the Twin Lakes 1971-1976

volumes of both lakes have remained at about 5 µl/l, a value expected of moderately eutrophic lakes (Vollenweider, 1968).

In 1976, the year after alum treatment, spring cell volumes were very different. West Twin was clear and had no blue-green algae, whereas East Twin was a greenish-brown color and supported a dense bloom of Oscillatoria rubescens. The same pattern occurred in spring 1977. By summer 1976 both lakes had a smaller cell volume than had occurred in any previous year, except 1973. Since algicide use was small in 1976, it appears that the alum application, which resulted in a lower epilimnetic total phosphorus but not SRP concentration, combined with the lower phosphorus income of 1976, was effective in lowering cell volume. It is suggested that the absence of a spring blue-green algae bloom in West Twin may also be due to the layer of aluminum hydroxide, since blue-green algae blooms apparently originate from populations in the sediments (Reynolds and Walsby, 1976).

The pattern of changes in surface chlorophyll of the lakes closely paralleled cell volume changes. Chlorophyll in West Twin dropped from a pre-diversion high in 1971 to the level of East Twin in 1972 (Figure 23). Both lakes were very low in 1976, as expected from the strong correlation between spring total phosphorus concentration and mean summer chlorophyll (Figure 5). Sakamoto (1966) suggested that eutrophic lakes will have a range of chlorophyll of 5-140 mg Chl a/M^3 .These lakes have had a summer monthly average of about 15 mg/M^3 since diversion and alum treatment, a level found in moderately eutrophic lakes.

Changes in the species composition of the summer surface phytoplankton followed a similar pattern each year. An early spring diatom bloom (Fragilaria, Asterionella) was followed by a sharp decline in cell volume in late April early May, and then a series of blue-green algae blooms occurred over the summer, with diatoms as sub-dominants during June and July. By August 90% or more of the cell volume was due to blue-green algae, either Anabaena limnetica or Aphanizomenon flos-aquae, and this persisted until November. The summer dominant in both lakes in all years except 1974 and 1975 was A. flos-aquae. A. limnetica was the 1974-75 dominant in both lakes.

In 1974 and 1975, Chlorophyta and Dinophyceae began to be a more abundant component of the late summer algae (Table 22). In 1976, the year after alum treatment, blue-green algae dominance was greatly reduced, particularly in East Twin, the downstream control lake.

The seston content of West Twin's surface waters was very high in 1971, but as with cell volume and chlorophyll, declined in 1972 and remained at this level through 1976 (Figure 24).

Changes in Secchi disc transparency were similar to those of cell volume, seston, and chlorophyll (Figure 25). In 1975, 1976, and also in 1977 (not shown) the spring (April-May) transparency of West Twin exceeded 4 meters while East Twin remained murky due to Oscillatoria. In 1976 the interval of high water clarity in West Twin persisted into July. The midsummer transparency has remained at about 2 meters following an initial increase between 1971 and 1972 for West Twin, a level found in moderately eutrophic lakes.

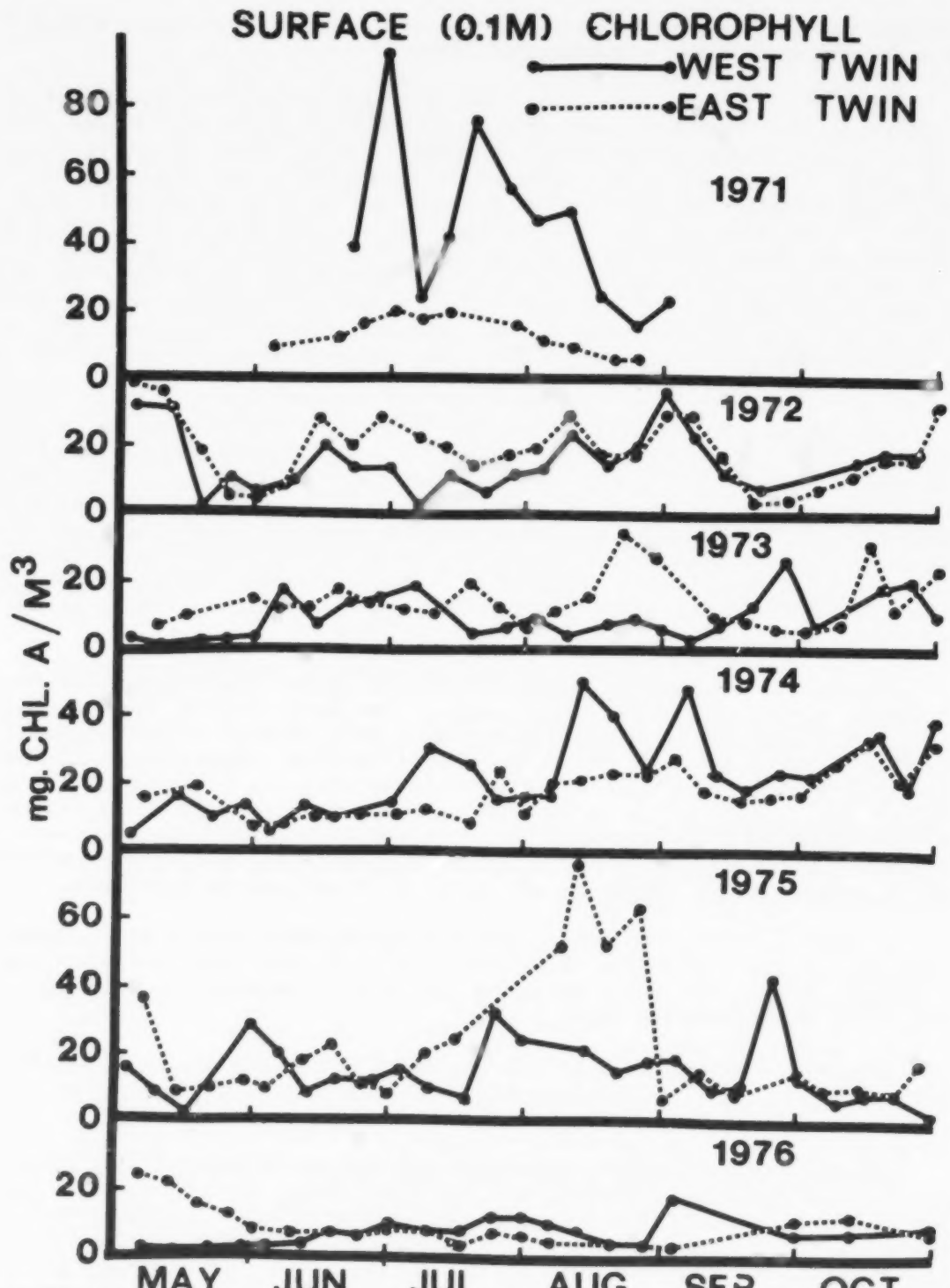

FIGURE 23. Changes in summer surface chlorophyll a of the Twin Lakes, 1971-1976

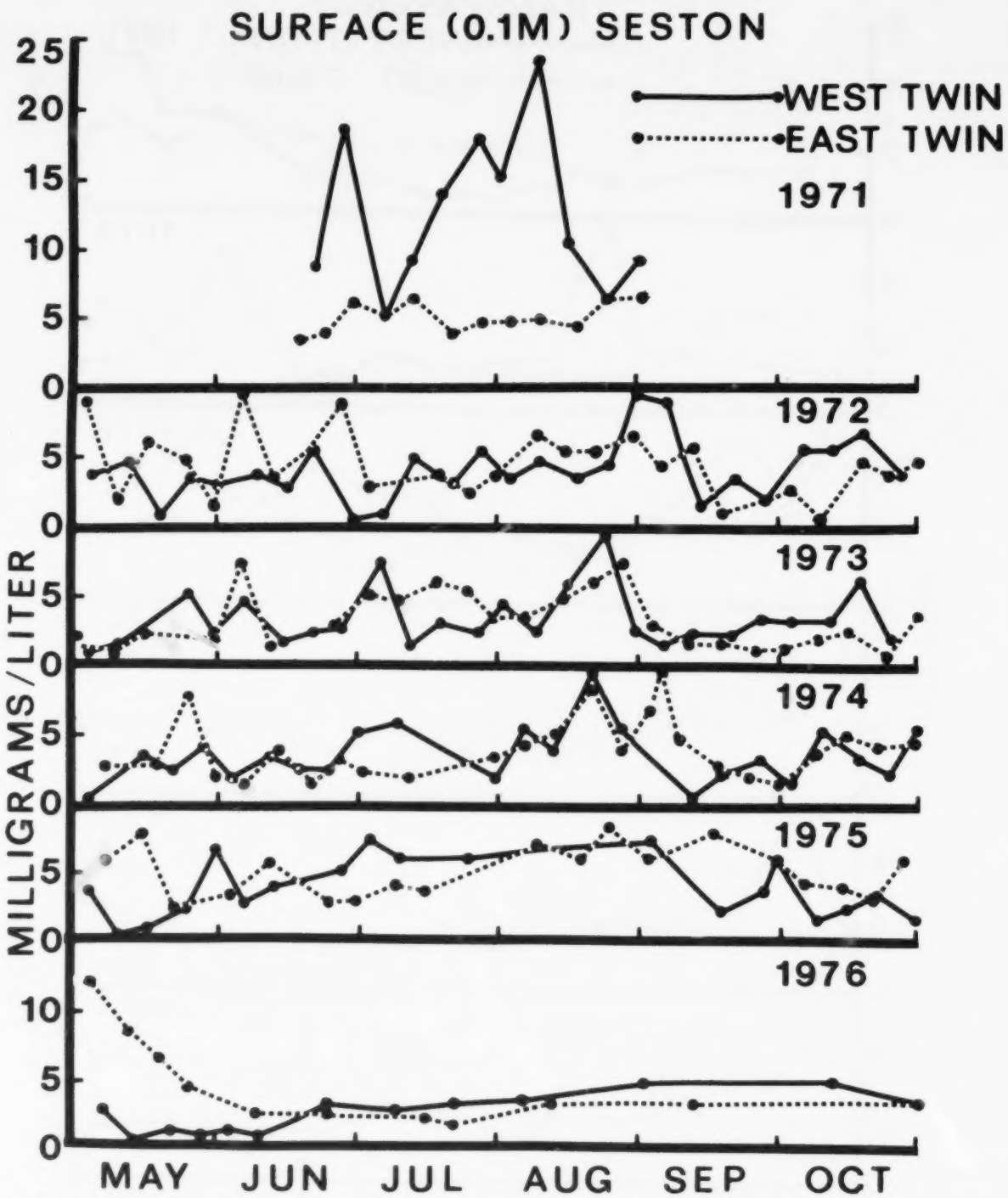

FIGURE 24. Changes in summer surface seston of the Twin Lakes 1971-1976

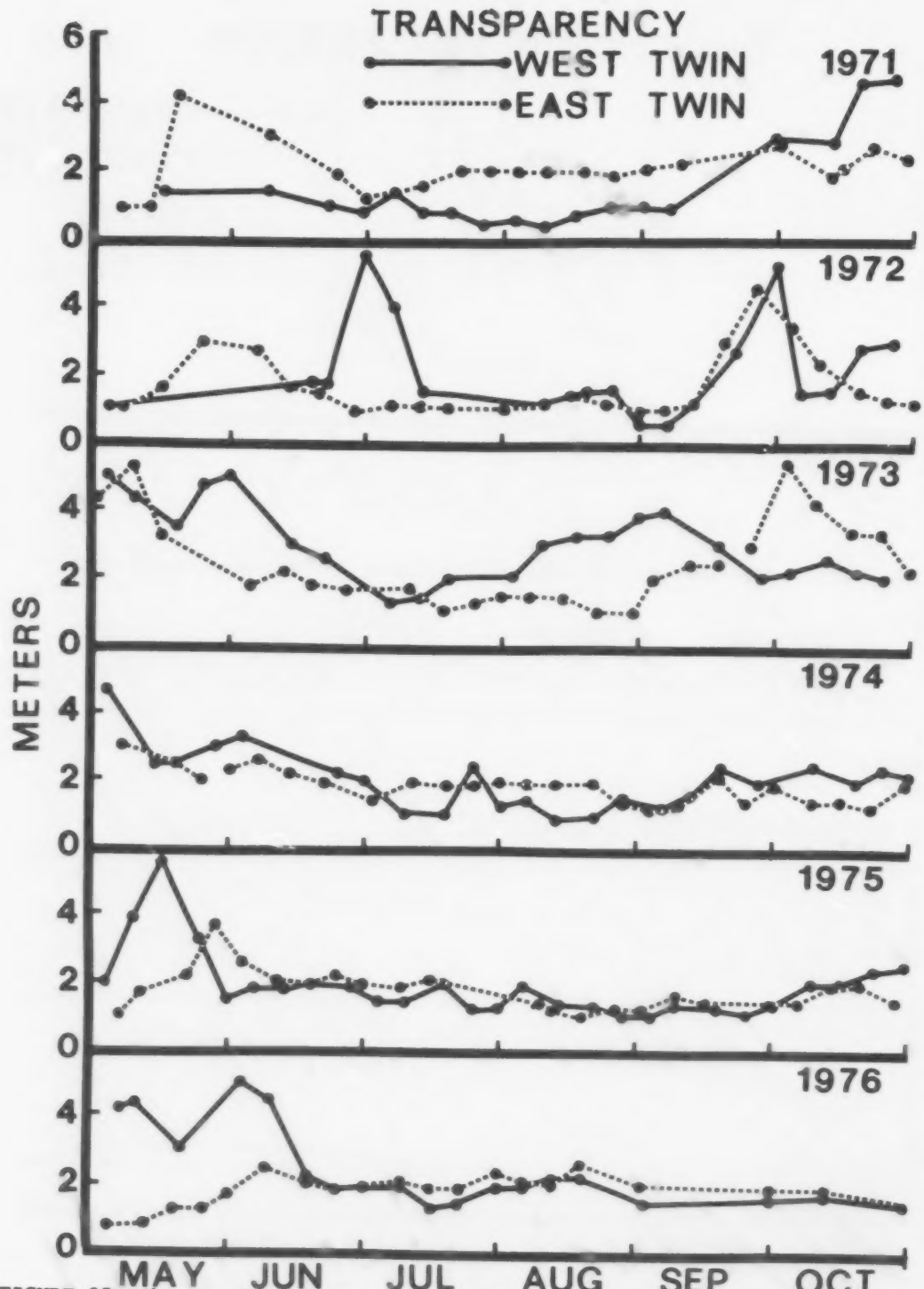

FIGURE 25. Changes in summer transparency of the Twin Lakes, 1971-1976

Table 22. MEAN PERCENT PHYLA AND SPECIES COMPOSITION OF AUGUST SURFACE (0.1 M) SAMPLES*

	West Twin					
	1971	1972	1973	1974	1975	1976
1. Blue-Greens	99.9(1)	98.1(2)	39.5(2)	93.8(1)	97.9(1)	84.8(1)
2. Diatoms	0.1	1.8(2)	43.3(2)	3.8(2)	0.8(2)	2.3(3)
3. Chlorophyta	0	0.1(1)	17.2(1)	0.2(1)	1.1(1)	0.1(1)
4. Dinophyceae	0	0	0	2.4(1)	0.2(1)	12.9(1)
Mean Cell Volume (µl/l)	17.88	3.27	0.12	4.68	5.24	2.16
	East Twin					
	1971	1972	1973	1974	1975	1976
1. Blue-Greens	88.8(1)	99.2(2)	97.5(2)	88.9(1)	97.0(1)	54.6(2)
2. Diatoms	11.3(1)	0.2(2)	2.5(1)	8.8(1)	1.0(1)	40.0(3)
3. Chlorophyta	0	0	0	0	0	0.5(1)
4. Dinophyceae	0	0.7(1)	0	0	1.9(1)	4.9(1)
Mean Cell Volume (µl/l)	3.03	4.24	7.00	3.84	7.88	1.76

Dominant Species

Blue-Greens

1. Anabaena limnetica
2. Aphanizomenon flos-aquae

Diatoms

1. Stephanodiscus
2. Cyclotella
3. Fragilaria

Chlorophyta

1. Sphaerocystis

Dinophyceae

1. Ceratium

*Numbers in parentheses refer to dominant species

Oxygen Deficits

There was no pattern of change in the oxygen deficits (Table 23) which could be directly related to septic tank diversion or to the alum treatment. Instead, as shown by Stewart (1976) for Lake Mendota, the deficits were positively correlated with day of ice-out (r = .797), suggesting that the oxygen deficit in lakes, once they are eutrophic, is not directly related to productivity.

Table 23. OXYGEN DEFICITS OF EAST AND WEST TWIN LAKES mg O_2/CM^2/Month

	WEST TWIN	EAST TWIN
1970*	1.60	1.22
1971	N.D.	2.25
1972	2.31	3.36
1973	2.29	1.56
1974	0.76	1.00
1975	2.37	2.64
1976	1.01	1.61

*Data from Heinz (1971)

Macrophytes

The distribution and biomass of macrophytes were surveyed by Rogers (1974), but have not been studied quantitatively after that. These plants remain as the major nuisance to most users of the lakes, and their abundance does not appear to have changed since diversion or alum treatment.

TROPHIC STATE INDEX

Many attempts have been made to index or express the eutrophic state numerically (and thereby objectively) by measuring and accounting for certain factors used qualitatively to identify a eutrophic lake. The most recent of such indices is the "Trophic State Index" (TSI) of Carlson (1977).

Eutrophic lakes are often described as having low transparency, high chlorophyll contents, and large quantities of total phosphorus. The Carlson TSI was calculated for each year from 1971, during the growing season (May 1 until October 31) for both lakes, from measurements of Secchi depth (SD), chlorophyll a uncorrected for phaeophytin (Chl A), and total phosphorus (TP), in samples drawn from the 0.1 M stratum. Additionally, a pre-diversion TSI was calculated using the transparency data of Heinz (1971) for 1969. Averages of TSI determinations made throughout the growing season are reported in Table 24.

The number of determinations is reported in the column designated n; in general these averages represented 20 or more observations per summer.

Table 24. TROPHIC STATE INDEX

	EAST TWIN LAKE			
	SD	TP	CHL *a*	N
1969	52.06	-	-	22
1971	52.21	47.88	56.02	9
1972	53.83	55.15	58.35	24
1973	49.43	55.41	55.76	21
1974	51.25	57.13	58.57	25
1975	52.85	55.47	58.40	25
1976	52.20	55.20	51.88	20
Mean	51.98	54.37	56.50	
	WEST TWIN LAKE			
	SD	TP	CHL *a*	N
1969	50.01	-	-	23
1971	63.40	57.09	66.87	11
1972	52.13	62.55	58.01	14
1973	44.51	60.91	50.66	20
1974	51.07	60.01	59.97	22
1975	52.04	54.18	55.98	31
1976	48.15	50.79	47.00	22
Mean	51.62	57.59	56.42	

The similarity of the TSI determined from total phosphorus and the TSI determined from Secchi depth and chlorophyll *a* indicate that the lakes were phosphorus-limited, supporting earlier considerations to that effect. These determinations also indicate that throughout the study interval both East and West Twin appeared to be moderately eutrophic lakes, as designated by Carlson (e.g., TSI of 50), and did not change from this category despite septic tank diversion. In this regard it should be noted that Vollenweider's boundaries of 10 µg P/l and 20 µg P/l as total phosphorus imply a TSI of 37.37 and 47.37, respectively. East and West Twin exceeded the upper boundary every year, including 1976, the post-alum treatment year.

The TSI for East Twin did not vary much from one year to the next, and for the years 1969 to 1975 the same can be said of West Twin, with the exception of 1973 determinations. In 1973 the TSI determination from Secchi depth and from chlorophyll a observations are substantially lower than other years. That the 1973 TSI determined from total phosphorus is not different from 1972 or 1974, suggests that the phytoplankton community was limited by some factor other than phosphorus, probably a very heavy addition of herbicides. The average TSI determined from each set of observations in 1976 was lower than previous years. Although this decrease is not statistically significant (one-way analysis of variance) it does suggest that the application of aluminum sulfate has had some impact on the degree of eutrophy. This view is supported by the reduction in cell volume and the change in algal species composition.

SECTION 7

DISCUSSION

The results of this study have implications for lake management, for lake classification and for general understanding of the eutrophication problem in small lakes.

Septic tank systems have been commonly installed in residential developments around lakes, and probably will be the usual method for domestic waste disposal in most future lake developments. This study has shown that these systems, even when properly installed in adequate soils, may not operate as designed and will discharge materials to the ground surface.

This finding is significant because it has been commonly believed that phosphorus will not enter lakes via groundwater in large amounts because of the sorptive capacity of soils. As earlier pointed out by Dudley and Stephenson (1973), old systems or those with high or perched water tables in coarse sandy soil may be expected to contaminate groundwater with phosphorus. The present study further supports the belief that septic tank systems should not be used near lakes because the leach fields appear to plug, inhibiting downward movement of water. The continued water income to the field from the home apparently forces water and nutrients to the ground surface. This is the cause of the commonly observed soggy lawn and resulted in very high concentrations in streams draining urban areas prior to diversion at Twin Lakes.

The diversion of the septic tank system alone did not result in a reduction of phosphorus income to the Vollenweider (1976) "acceptable" level. This was confirmed by levels of algal standing crop five years after diversion and by a Trophic State Index (Carlson, 1977).

This response to diversion was to be expected. The watershed was developed as a residential-recreation area away from the usual urban setting of industry, traffic, and congestion. However, development occurred without plan so that erosion and street wastes as well as seeps from septic systems which were placed near the lake and not maintained, added to the natural richness of the water and also accelerated the filling of the lake. The result was lakes that do not provide the complete recreational satisfaction envisioned by either the developers or the residents.

Unlike lakes which receive point sources of nutrients, these lakes have become culturally eutrophic from diffuse sources. Development continues on the watershed, in many cases on lots which are not suited for homes unless the land is completely cleared, a marsh filled, or where some other operation is required which has a large impact on the lakes. The usual development

practice is to leave the cleared land without cover for a year or more during home construction, thus adding tons of soil to the lake. As a result of years of these practices, the littoral zone has expanded from erosion, creating a favorable environment for the major nuisance, macrophytes.

It is not likely that this very common type of eutrophication problem will be solved by reducing the income of nutrients alone through septic tank diversion. A rigorous land use policy and storm water diversion might slow the further deterioration of the lakes, but these steps are expensive and politically unpopular, and in some cases impossible if the land owner has the deeded right to develop the land, regardless of its suitability for such development. Even if phosphorus income can be greatly reduced, this may not alter the macrophyte problem and these plants and their associated biota may continue to add phosphorus to the epilimnion at rates sufficient to support algae growth, even after further nutrient diversion.

The prognosis for the Twin Lakes, and the many hundreds of lakes like them, is not favorable unless steps are taken to manage the land in ways which have less impact on the lakes. Restoration of such lakes then might be possible, although the cost could be high.

The diversion of septic tank drainage at Twin Lakes, however, was essential to prevent further deterioration of water quality and to return the lakes to a condition more acceptable to lake shore residents. Based upon measurements in 1971 and early 1972, before diversion, the pre-diversion phosphorus income may have been more than twice as high as the immediate post-diversion income and would have continued to climb had there been no diversion, due to leach field failures and continued planless construction.

This study has provided evidence, from the hypolimnetic aluminum sulfate application to West Twin, that the littoral zone may be a very significant internal source of phosphorus. The source(s) within the littoral was not identified. This suggests the hypothesis that there may be two classes of eutrophic lakes. The first class, like the Twin Lakes, has small pelagic-littoral area ratios and internal loading is dominated by sources such as macrophytes-epiphytes, fish and groundwater. Lakes of this class are likely to be small in total area and perhaps protected from wind by features of the landscape, thus reducing the possibility of large internal seiches. The second class has a large pelagic-littoral area ratio, is usually much more exposed to winds and thus has large seiches, and vertical entrainment of phosphorus from the hypolimnion is the main internal source of phosphorus. Lake Mendota as described by Stauffer and Lee (1973) may be of this type, although the littoral of that lake is extensive and must be important.

Based upon this hypothesis and the observations of internal loading after alum treatment of the Twin Lakes, hypolimnetic application might be most effective on large lakes. However, cost and ease of application might be prohibitive.

The hypolimnetic aluminum sulfate application may continue to reduce the phosphorus concentration in West Twin since sedimented material, as well as

that which might have been released from hypolimnetic muds, appears to be satisfactorily controlled. This may lead to smaller summer algal blooms if spring concentrations remain low.

One of the most important areas for further research, based upon the results of this investigation, will be to identify and quantify the sources of internal phosphorus loading from the littoral, and to assess the significance of groundwater in moving phosphorus into the water of the littoral zone as it passes through phosphorus-rich sediment. The next alum application should be in the spring to the littoral area of a small lake for which there is a good phosphorus budget.

The preponderance of research in eutrophication has been directed toward algal problems, and these studies have correctly identified this nuisance as the most significant in large lakes, particularly those which deserve protection due to aesthetic, recreational, or commercial reasons. However, of equal significance is the much smaller residential or real estate lake development. In many of these systems, particularly throughout the Midwest and the Southeastern states, nuisance macrophytes pose an equally difficult and important problem. This is true at the Twin Lakes. Not only do these highly productive littoral zones interfere with many types of water use, they may also release sufficient nutrients to the open water to sustain algal blooms after a significant diversion of external income.

As previously noted by Odum (1969), Patten (1973), and Cooke (1973), understanding and control of the eutrophication problem requires an ecosystem perspective rather than the narrower nutrient concentration - algal biomass approach. At Twin Lakes, for example, long-time residents have observed that the lakes became much shallower after intense development started. This probably provided a greatly increased littoral zone with macrophyte development. While summer plankton chlorophyll in the Twin Lakes is strongly related to ice-out phosphorus concentration, and the ice-out concentration can be predicted from a loading model, this research has also indicated that phosphorus present in the spring is released to the water column in the summer by both biotic and/or abiotic components of the littoral zone. Thus control of algal blooms at Twin Lakes will involve not only a reduction in nutrient income through diversion of domestic wastes, but land management and control of the internal nutrient sources.

The eutrophication problem for smaller lakes appears to be more complex than for larger lakes because more components of the lake-watershed system are significantly involved. A solution to the problem for smaller lakes, however, may be more likely since watershed residents are more intimately involved in water use and because the political-economic base is local rather than regional.

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APPENDIX

INTRODUCTION

A general outline of our approach to aluminum sulfate dose determination, experiments to test its effectiveness, and application procedures was given in the Materials and Methods section. Here the step-by-step approach taken during this investigation is described in detail. Experiments were confined at first to the laboratory, then to in situ enclosures to test toxicity and effectiveness, and finally to a pilot lake treatment (Dollar Lake). It was believed that if, at any step, there was evidence of the possibility of damaging side effects then no full-scale lake treatment would occur. The successful treatment of the eutrophic Dollar Lake (1974) was the final step before treatment of West Twin in 1975. A further description of these experiments, including a complete description of the Dollar Lake treatment, is given in Wilbur (1974) and Kennedy (1978).

The Appendix is organized to give the reader a summary of the experiments. The procedure for the determination of West Twin's dose is given in detail. Finally, a section is presented on the equipment and application design, the application procedure for the West Twin treatment, and a summary of costs. Non-metric units are used in portions of the Appendix since aluminum sulfate is supplied as gallons or pounds and temperature and other corrections are based on these units.

DOSE DETERMINATION EXPERIMENT

Lake waters collected at varying depths (and thus with varying total alkalinity) in West Twin Lake and Dollar Lake were treated with aluminum sulfate at dosages ranging from 0-50 mg Al/l to confirm the relationship between the amount of aluminum added and initial alkalinity. One liter jars containing 500 ml of lake water were stirred at 40 RPM for two minutes following the addition of a liquid concentrate of aluminum sulfate. Changes in pH were measured during the first two minutes. Supernatants were subsampled following a 12 to 24 hour settling period and measurements of alkalinity and dissolved aluminum taken.

Results obtained from the treatment of West Twin Lake 7 meter water (initial total alkalinity = 137.3 mg $CaCO_3$/l) are typical (Figure A-1). Doses of 1 to 26 mg Al/l produced a large settleable floc, while doses above 26 mg Al/l resulted in a white turbid reaction mixture which did not settle after 18 hours. No measurable floc was produced at dosages below 1 mg Al/l. Alkalinity dropped rapidly between 0.5 and 26 mg Al/l (137.3 to 6.0 mg $CaCO_3$/l) and was completely removed above a dose of 30 mg Al/l. pH was reduced from 7.2 to 5.13 at 26 mg Al/l and to 4.9 at 30 mg Al/l.

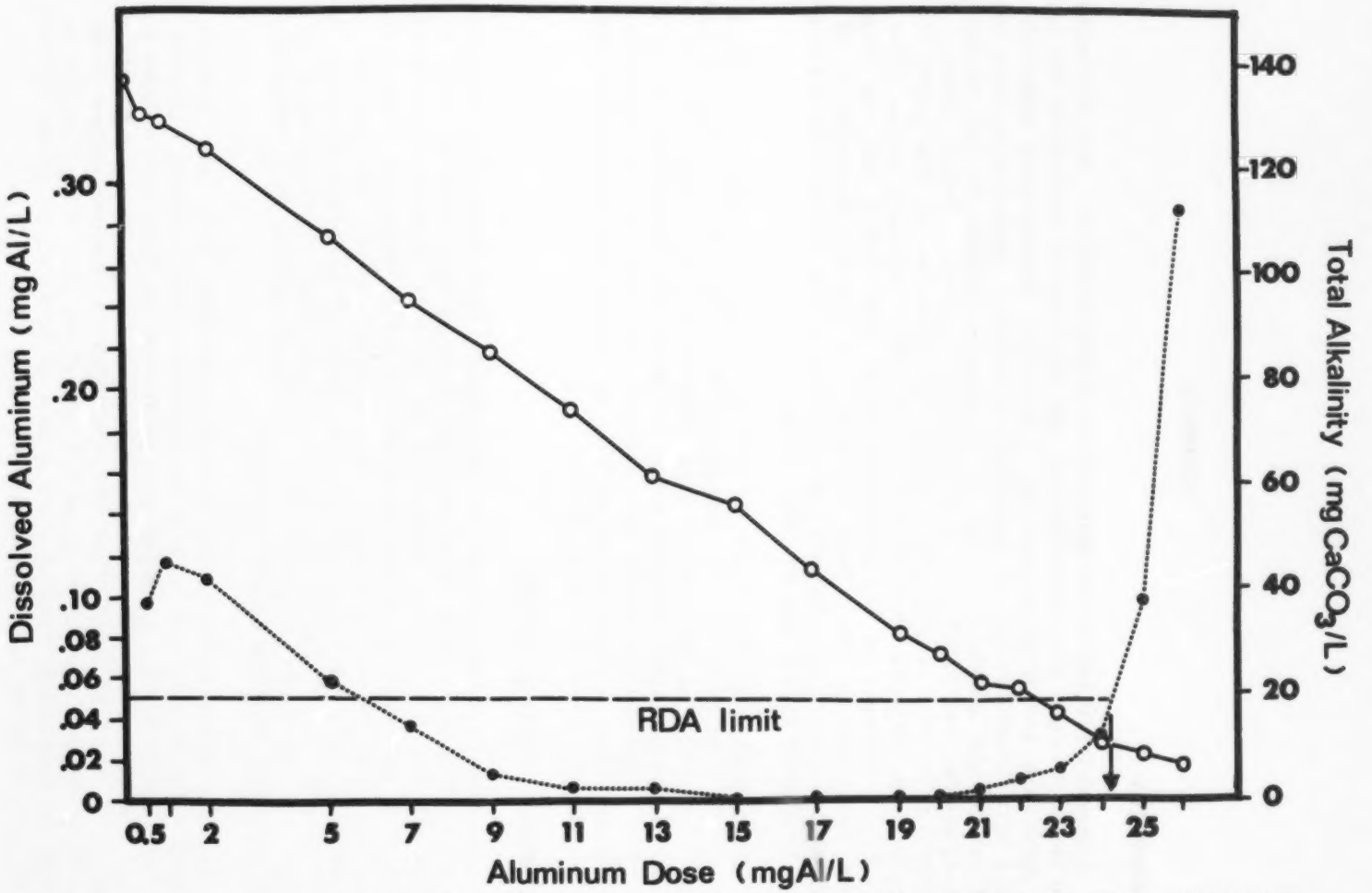

FIGURE A-1. Relationships of aluminum dose, residual dissolved aluminum, and lake water alkalinity

Residual dissolved aluminum, high in the dose range 1 to 7 mg Al/l, remained low (.02 mg Al/l) in the dose range 9 to 23 mg Al/l. Doses above 24 mg Al/l resulted in elevated residual dissolved aluminum concentrations. The maximum dose for this alkalinity was calculated to be 24.2 mg Al/l since this dosage represents the maximum amount of aluminum which can be added (under the conditions of this test, i.e., 2 minute mixing) before residual dissolved aluminum concentration exceeds 0.05 mg Al/l, a level reported to be tolerated by trout (Everhart and Freeman, 1973).

Maximum dosages calculated in a like manner for the full range of alkalinities found in these lakes were used to determine the relationship between alkalinity and aluminum dose using linear regression techniques. The relationship is expressed as

$$y = -3.4460 + 0.1950X$$
$$r = .967$$

Where:

y = maximum dose (mg Al/l)
x = total alkalinity (mg $CaCO_3$/l)

The slope of this line was compared to that based on stoichiometry and was found not to be significantly different ($P < .05$). The low intercept value, as yet unexplained, suggests the necessity for performing such tests prior to any lake treatment to insure low residual dissolved aluminum values and to avoid unnecessarily low pH. The suggested procedure to be followed in the calculation of maximum dose to stratified lakes is to determine the maximum doses to samples of varying alkalinity collected at several depths in the water column. The depths chosen should span the range of alkalinities found in the lake. These maximum doses can then be plotted versus initial alkalinity (e.g., Figure A-2) with the resulting linear relationship being used to determine maximum dose at any alkalinity.

PHOSPHORUS REMOVAL EFFICIENCY

Phosphorus removal efficiency was evaluated by treating lake water spiked with inorganic phosphorus (final concentrations ranging from 53 µg P/l to 1153 µg P/l) with a dose of 15.14 mg Al/l, the maximum dose for water with an alkalinity of 97.3 mg $CaCO_3$ /l. Following settling, final total phosphorus concentrations ranged from 15 to 52 µg P/l indicating that efficient removal could be achieved over the range of phosphorus concentration commonly experienced in these lakes during summer stratification. Phosphorus concentration did not appear to interfere with floc formation.

A second experiment was designed to determine the maximum phosphorus retention capacity of the aluminum hydroxide floc and thus its potential for retarding the upward movement of soluble phosphorus at the mud-water interface. Four elution columns were constructed of 50 cc disposable syringes. The barrels of the syringes were inverted and the bottom 2 cm packed with glass wool to prevent the loss of floc material. Three columns received a known quantity of pre-formed floc, whose aluminum content ranged from 9.957 mg to 8.959 mg. The fourth column contained only glass wool as a control. Each

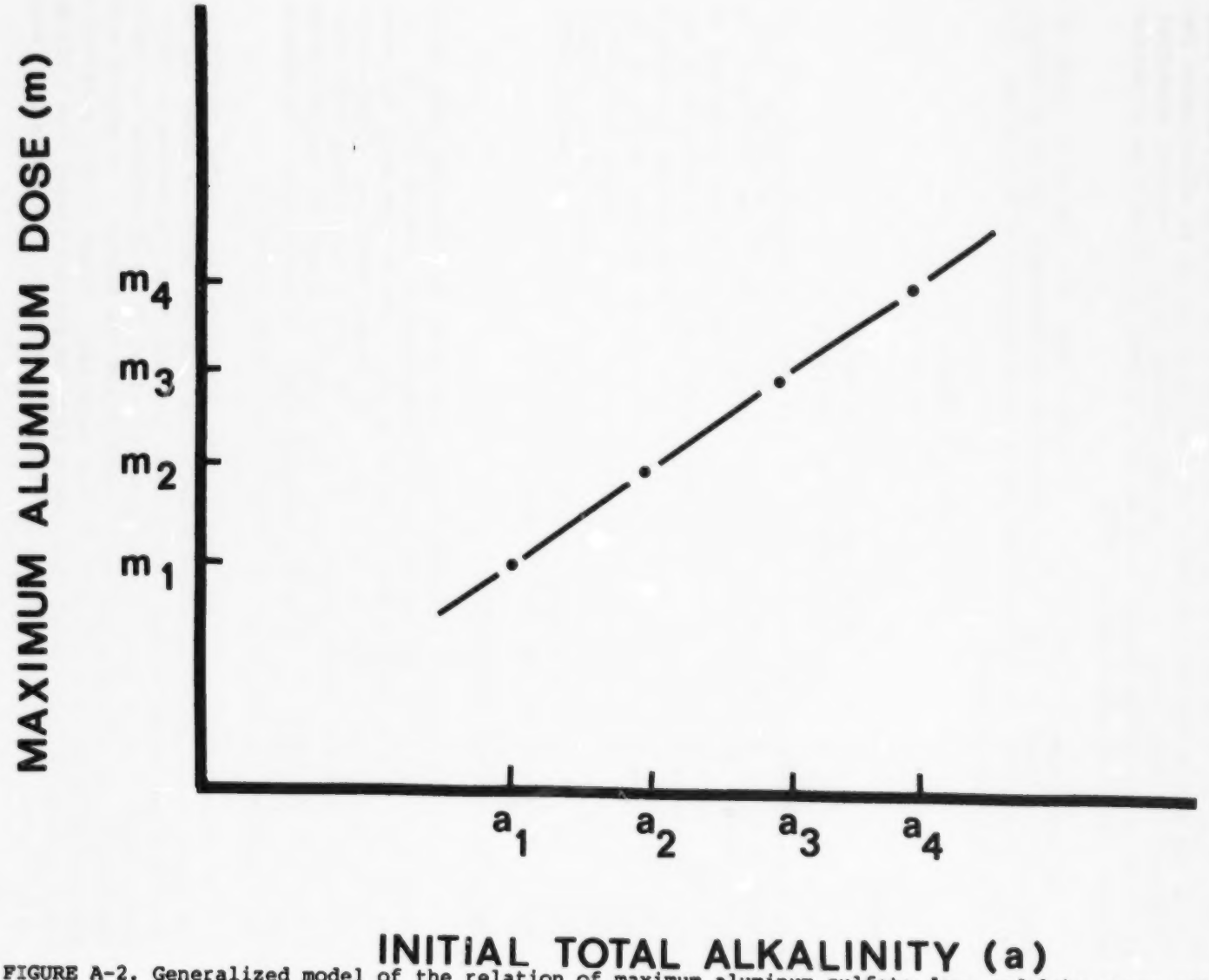

FIGURE A-2. Generalized model of the relation of maximum aluminum sulfate dose and lake water alkalinity

column was capped and fitted with inflow and outflow ports and eluted at a rate of 8.31 ml/h with a phosphorus solution containing 10 mg P/l. The pH of the three experimental columns was maintained at pH 6, 7 and 8 to bracket the pH values which might be found at the mud-water interface. Fractions were collected at 100 ml intervals for phosphorus determinations. The amount of phosphorus lost to the columns was calculated by difference.

The maximum phosphorus removal capacity at pH 6, 7 and 8, under the conditions of the test, was reached at Al to P molar ratios of 1.863, 2.589, and 3.311, respectively. There was no significant ($P < .01$) loss of aluminum from the columns.

This experiment provided further evidence to support the belief that dosage should be based upon water alkalinity and not upon the phosphorus content of the lake. The object of the treatment is to retard phosphorus release from lake sediments. To do this effectively and over long periods requires the largest amount of aluminum on the sediments, consistent with an upper limit based on RDA. The effectiveness of the aluminum is related to pH of the sediments. Eutrophic lakes with low alkalinity and with high interstitial water pH cannot be given a large dose and the floc will be less effective than in the opposite situation.

IN SITU COLUMN EXPERIMENT

In situ experiments were conducted in West Twin Lake in summer 1973 to ascertain the effectiveness and longevity of treatment under field conditions. Six polyethylene columns, open at the top to air and at the bottom to sediment, were used to isolate water columns 8 meters long. The columns had a diameter of 1.3 M, were anchored in the loose sediments with a ring of ½ inch steel rod, and were supported above water level by styrofoam floats. A dose of 485.6 gm of aluminum sulfate (the sum of maximum doses to each one meter interval from 5 M to the bottom) was added to the hypolimnion of three of the columns as a liquid concentrate. Three columns were held as controls. Mixing (in both experimental and control columns), accomplished with a hand-held, long-handled horizontal paddle, lasted 2 minutes and was restricted to the volume of water between 5 and 6 M to simulate conditions of a lake-scale treatment employing a horizontal applicator manifold.

Water samples were collected at 0, 1, 2, 5 and 7 meter depths on alternate days during the first 14 days. On day 18 the columns were sealed at 5 M by constricting the columns around tube which extended from the 7 M depth to the surface, thus allowing the closed columns to be sampled. Previous experience with columns (Kennedy, 1978) indicated that algae growths on the column wall occur in the lighted zone and will bias samples taken in upper waters. Samples were then obtained for 86 days. The total duration of the experiment was 104 days, during which time the hypolimnion in the columns remained anoxic.

Floc formation and sedimentation occurred rapidly with only traces of floc detectable 10 hours after treatment. Hypolimnetic waters drawn from experimental columns were noticeably clearer than those from controls.

Maximal soluble reactive phosphorus removal occurred during the first hour, with mean concentrations at 5 M and 7 M dropping from 68 µg P/l to 7 µg P/l and 288 µg P/l to 8 µg P/l, respectively. Maximum reduction in total phosphorus concentration was not completed until 10 hours after treatment, at which time the mean concentrations at 5 and 7 meters were 33 µg P/l and 43 µ g P/l, respectively. Filterable total phopshorus concentrations at 5 and 7 M were also reduced during the first 10 hours. There was a small reduction in particulate phosphorus, which was probably the result of continued sedimentation of seston from upper waters.

Visual inspection of the sediments in the treated enclosures by SCUBA divers at the time of closure (day 18) indicated a 6 to 8 cm floc layer covering sediments. Manipulation of the enclosures during closing had no apparent effect on water chemistry since no differences existed between 7 M samples withdrawn just prior to and just following closing.

Continued monitoring of the 7 M sampling ports showed that the application of a maximum dose of aluminum sulfate retarded sediment phosphorus release for extended periods of time. Mean total phosphorus concentration at 7 M in the controls increased from a low of 298 µg P/l on day 32 to 390 µg P/l at the termination of the experiment. Mean total phosphorus concentration in the experimental columns increased from 18 µg P/l on day 18 to 30 µg P/l at termination. Since the bottom sediments represented the only source of phosphorus for the enclosures, rates of change in concentrations can be expressed as sediment release rates. The sediment phosphorus release rate in the controls was 3.69 µg P/M^2/day as opposed to 0.30 µg P/M^2/day in the experimentals. This represents a decrease of 91.9% as the result of treatment, leading to the conclusion that the aluminum hydroxide layer is effective in controlling sediment phosphorus sources under anoxic conditions.

As predicted, pH and total alkalinity were reduced immediately following treatment. Alkalinity remained low throughout the experiment, but pH recovered to near control values by day 50. Residual dissolved aluminum levels, although high initially, never exceeded the 0.05 mg Al/l upper limit.

ALUMINUM TOXICITY EXPERIMENT

Experiments were conducted to determine whether or not the application of a maximum dose of aluminum sulfate would result in toxic side-effects and included both laboratory and field studies (Wilbur, 1974).

Laboratory bioassays, employing Pimephales promelas (Northern Fathead minnow), were conducted in 56.85 liter (15 gallon) aquaria which received varying amounts of aluminum sulfate and which were maintained at varying pH's. Aluminum sulfate dosages ranged from 0 mg Al/l (control) to 42.0 mg Al/l, bracketing the range of maximal doses required to treat Dollar and West Twin Lake. pH was maintained at 8.2, 7.0, 6.2 and 4.9 in separate aquaria at each dose level. Resultant average dissolved aluminum concentrations ranged from 0.009 mg Al/l to 10.10 mg Al/l. Twenty-three to thirty-four fish (1½" to 3" in length) were introduced into the aquaria immediately following treatment with percent mortality calculated over a two week period.

Complete (100%) mortality recorded after one week occurred in aquaria maintained at pH 4.9 with doses of 10.5, 21.0 and 42.0 mg Al/l (Table A-1), and appeared to be related to elevated dissolved aluminum concentrations (1.56, 1.80, and 10.10 mg Al/l, respectively). High doses to waters maintained at pH's above 6.2 and waters maintained at pH's down to pH 4.9 and receiving no aluminum sulfate, resulted in minimal (8%) mortality. Based on the response of this species, maximal doses of aluminum sulfate, which by our definition result in dissolved aluminum concentrations equal to or less than 0.05 mg Al/l, should not result in significant toxicity problems to fish.

Table A-1. PERCENT MORTALITY OF Pimephales promelas AFTER ONE WEEK

	Treatment Group			
pH	Control (0.018 mg Al/l)	Low (10.5 mg Al/l)	Medium (21.0 mg Al/l)	High (42.0 mg Al/l)
8.2	0.0 n=24	0.0 n=27	0.0 n=30	7.7 n=26
7.0	0.0 n=24	0.0 n=30	0.0 n=22	0.0 n=23
6.2	0.0 n=29	36.0 n=28	0.0 n=26	8.7 n=23
4.9	0.0 n=23	100.0 n=34	100.0 n=34	100.0 n=31

From Wilbur 1974

In situ enclosure experiments were conducted to determine the effects of aluminum sulfate application on benthic macroinvertebrates in littoral communities. Triplicate enclosures in East Twin Lake (0.5 M Depth) containing about 325 liters of lake water received one of four treatments:

Group A = Maximum dose of aluminum sulfate

Group B = Maximum dose of aluminum sulfate plus enough calcium carbonate to maintain pre-treatment pH.

Group C = Addition of sulfate (as sodium sulfate) equivalent to that added in treatment A and B.

Group D = Lake water only (control)

The total number of species and total number of individuals were determined one day following treatment and at the end of two weeks. Two indices, coefficient of community (CC) and percentage similarity of community (PS) were used to assess the effects of the above treatments on macrobenthic communities. Both indices measure similarity between two sampling quadrats or, as in

this case, two sampling dates. If the two have no species in common, both CC and PS will equal zero. The maximum value of CC (i.e., 100) occurs when both samples have identical species lists, and the maximum value of PS (i.e., 100) occurs if both samples have the same species in the same proportions (Pielou, 1975). Results of these analyses suggest that the application of aluminum sulfate, or its accompanying side-effects (high sulfate concentrations and low pH), does not seriously effect macrobenthic organisms or macrobenthic community structure. With the exception of the buffered-alum treatment enclosures, all enclosures experienced a similar decrease in numbers of organisms during the interval day one to day fourteen (Table A-2). All of the enclosures showed little change in total number of species (Table A-2). Comparisons of diversity (similarity) indices indicated a high affinity between sampling dates for all treatment groups (Table A-3).

Table A-2. BENTHIC MACROINVERTEBRATES OF GROUPS AFTER TREATMENT

Group	# of Organisms		# of Species	
	After One Day	After 2 Weeks	After One Day	After 2 Weeks
A (alum)	868	415	30	26
B (alum-buffer)	602	657	26	23
C (sulfate)	1007	463	26	28
D (control)	727	424	25	24

Table A-3. DIVERSITY INDEX VALUES COMPARING SAMPLING DATES OF EACH GROUP

Group	CC	PSc
A	55	69
B	63	56
C	68	74
D	58	73

From Wilbur 1974

DOLLAR LAKE TREATMENT

The results of these experiments with effectiveness and toxicity led to a pilot lake treatment. Dollar lake, a small (2.2 ha), dimictic, shallow (Z_{max} = 7.5 M) eutrophic lake north of East Twin was given a hypolimnetic treatment in July 1974. The complete results are described in Kennedy (1978). Two principal observations were made which formed the basis for the West Twin treatment. First, no toxicity to fish was seen. Secondly, the alum was very effective in removing phosphorus from the water column and in retarding release from the sediments.

An experiment was conducted to measure phosphorus release from treated and untreated sediments. Plastic wastebaskets, fitted with sampling ports to which a plastic bag could be attached, were inverted and placed in hypolimnetic muds before treatment. After treatment additional baskets were inverted over floc-covered sediments next to the control baskets. Samples were taken over a period of one week in 1974 and 1976 and the water analyzed for total phosphorus. Treated muds (Table A-4) released very small amounts of phosphorus, particularly in 1974, further substantiating the conclusions from the in situ columns that alum-treated anoxic sediments have low phosphorus release rates, and the conclusions that continued internal phosphorus loading in West Twin after treatment must come in part from a source other than anoxic hypolimnetic sediments.

Table A-4. TOTAL PHOSPHORUS CONCENTRATION IN WATERS FROM TREATED AND UNTREATED ANAEROBIC DOLLAR LAKE SEDIMENTS

Depth of Half-Barrels	1974 µg P/l	1976 µg P/l
6M untreated	701	1021
6M treated	130	224
4M untreated	315	200
4M treated	198	705
4M untreated	392	3519
4M treated	33	600

DOSE DETERMINATION FOR WEST TWIN LAKE TREATMENT

Preparatory to the treatment of West Twin Lake, a series of jar tests was conducted using water collected at 5, 7, 9 and 11 M to confirm the relationship between alkalinity and dosage. Treatment was to be confined to the hypolimnion (5 M and below).

As before, 500 ml subsamples of lake water from a given depth received increasing amounts of aluminum sulfate, were stirred for 2 minutes, and then allowed to settle for 24 hours. Each jar in each alkalinity series was then sampled and alkalinity and dissolved aluminum measurements made.

Total alkalinities in mg $CaCO_3$/l determined immediately prior to alum addition were:

Depth	Total Alkalinity
5 M	115.33 mg $CaCO_3$/l
7 M	137.33
9 M	143.93
11 M	149.00

With the exception of 7 M water, which received an aluminum dose ranging from 0.5 mg Al/l to 26 mg Al/l all depths received a range of doses intended to bracket the critical portion of the titration curve. Results of these jar-tests are given in Table A-5. In each case, dissolved aluminum concentrations increased exponentially as the critical concentration of 0.05 mg Al/l was approached and exceeded. Therefore, dissolved aluminum concentrations determined at each dose level in each alkalinity series were plotted versus dosage and fitted with an exponential curve in order to determine the maximum aluminum dose at each alkalinity:

$$y = ae^{bx}$$

where:

y = dissolved aluminum concentration (mg Al/l)

x = aluminum dose (mg Al/l)

Values for b and a determined for each data set were:

Depth	a	b	r
5 M	8.518×10^{-9}	0.7875	0.98
7 M	9.40×10^{-11}	0.8294	0.99
9 M	7.121×10^{-15}	1.1938	0.96
11 M	3.30×10^{-8}	0.8401	0.98

Table A-5. JAR TEST RESULTS-DISSOLVED ALUMINUM CONCENTRATION (mg Al/l)

Dose (mg Al/l)	5 meters	7 meters	9 meters	11 meters
17	0.007	0.0005	-	-
18	-	-	-	-
19	0.0166	0.0008	-	-
20	0.0702	-	-	-
21	0.1106	0.0040	-	-
22	0.3527	0.0097	-	-
23	-	0.0138	0.0040	0.0059
24	-	0.0311	0.0317	0.0286
25	-	0.0950	0.0850	0.0505
26	-	0.2850	0.1540	0.1055
27	-	-	-	0.2050

A dissolved aluminum concentration of 0.05 mg Al/l was substituted for y allowing the calcualtion of x, the maximum dose of aluminum (in mg Al/l) required at each alkalinity.

Depth	Total Alkalinity	Maximum Dose
5 M	115.33 (mg $CaCO_3$/l)	19.791 (mg Al/l)
7 M	137.33	24.221
9 M	143.93	24.778
11 M	149.00	25.167

The apparent relationship between alkalinity and maximum dosage was determined by subjecting these data to linear regression analysis.

$y = 0.849066 + 0.16597x$ ($r=0.985$)

where:

y = aluminum dose (mg Al/l)

x = initial total alkalinity (mg $CaCO_3$/l)

This relationship was then used to determine the maximum dose for all alkalinities within the range 115.33 to 149.0 mg $CaCO_3$/l.

The next step was to determine the dosage to be applied to each one meter thick strata. The alkalinity of each one meter depth from 5M to 11M was determined in triplicate, with the mean alkalinity of each one meter interval (e.g., 5 M to 6 M, 6 M to 7 M, etc.) obtained by averaging. Based on the above relationship and these average alkalinities, the (volumetric) maximum dose for each depth interval was calculated. Based on the formula weight of 594.19 (assuming $Al_2(SO_4)_3 \cdot 14H_2O$) these maximum doses were converted to pounds of dry alum per cubic meter of lake water by multiplying by 0.02428, the conversion factor to change mg Al/l to pounds alum/M^3 , as follows:

Depth Interval	Average Alkalinity	Max. Dose	lb Alum (dry)/M^3
5-6 M	121.35 mg $CaCo_3$/l	20.907 mg Al/l	0.5076
6-7 M	132.35	22.934	0.5568
7-bottom	142.91	24.880	0.6041

A dose expressed in terms of pounds of dry alum per cubic meter was sufficient to determine the total amount to be added to the lake if dry alum was to be used. However if liquid alum was to be used, as was the case for the treatment of West Twin Lake, further calculations were necessary to express dose in terms of gallons per cubic meter.

The standard strength of commercial alum liquors ranges from 8.0 to 8.5% Al_2O_3, which is equivalent to 5.16 to 5.57 pounds dry alum per gallon (at 60 F). Since liquid alum is shipped at temperatures considerably higher than 60°F, correction must be made to allow for the resulting lower density. For example, one gallon of 8.5% Al_2O_3 liquor will contain 5.57 pounds at 103 F. Alum suppliers provide results of Al_2O_3 assays with each shipment. It is important to note that such assays are reported as percent Al_2O_3 at a standard temperature of 60° F. The supplier should also be able to estimate the temperature of the liquor at the proposed time of application by taking transport and storage time into account. It is advisable that the liquor temperature be re-checked prior to treatment in case such an estimate is grossly in error. Knowledge of both temperature and Al_2O_3 is required in order to determine dosage in terms of gallons per cubic meter.

The following procedure was used to determine dosage in terms of gallons per cubic meter.

1. Percent Al_2O_3 (at 60° F) was converted to density, expressed as degree Baume (at 60° F) using Figure A-3. The liquor purchased for the West Twin Lake treatment assayed at 8.3% Al_2O_3, or 36.5° Be (60° F).

2. A temperature correction was applied against this Baume to account for the decrease in density using Figure A-4. The estimated temperature of the liquor at time of application was 100-105° F (a temperature of 103° F

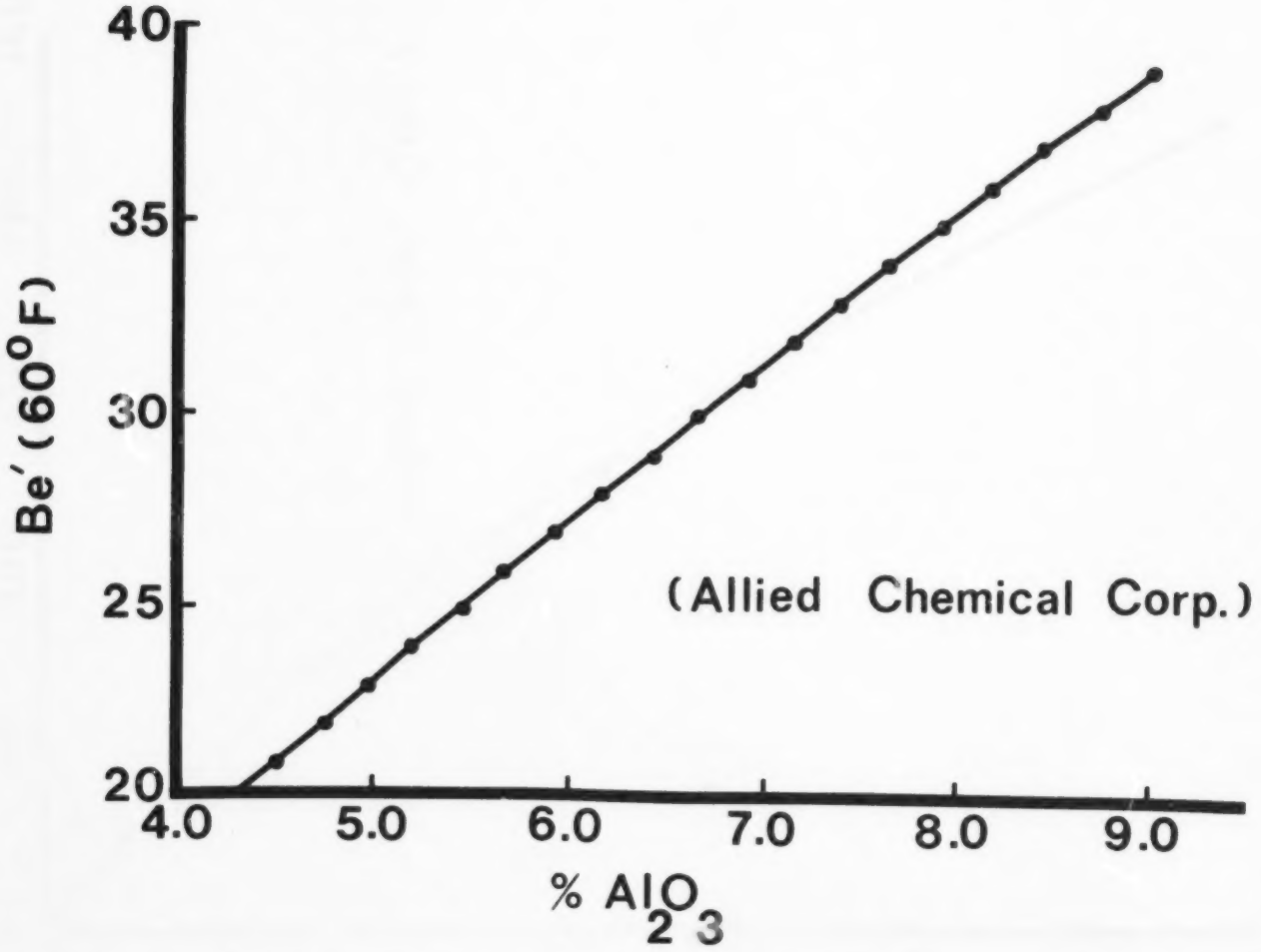

FIGURE A-3. Relationship of Baume (60°F) and percent $Al_2 0_3$ (Anonymous, 1970)

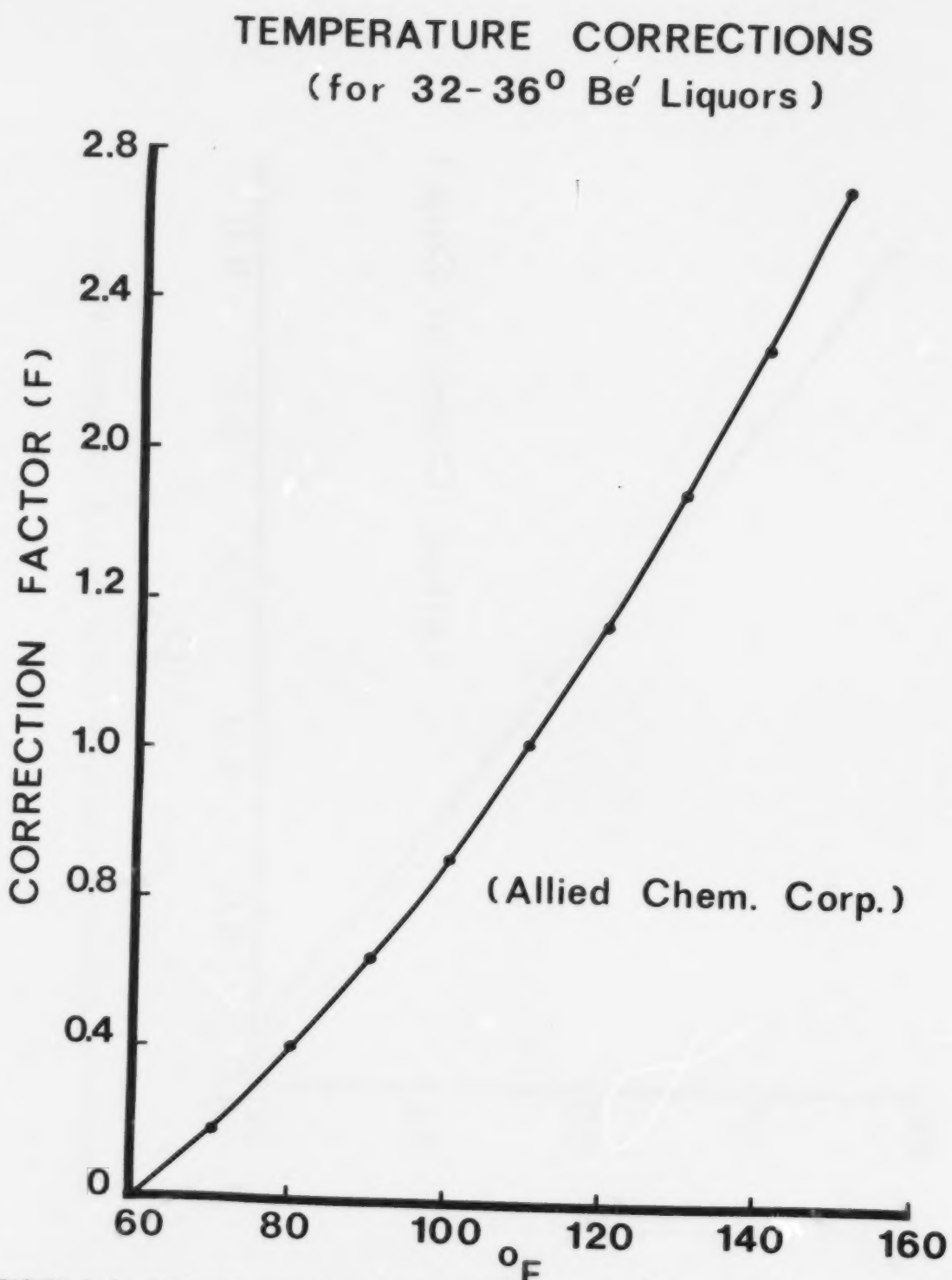

FIGURE A-4. Temperature correction factors for 32-36° Be' liquors (Anonymous, 1970)

was used). The corresponding correction factor (for 32° -36° Be liquors) (Anonymous, 1970) was 0.905. The adjusted Be was determined by subtracting this factor from °Be (60° F).

°Be (60° F)-F = adjusted °Be

The adjusted °Be was then 35.595 °Be.

3. Pounds of alum per gallon were next determined on the basis of the adjusted °Be using Figure A-5. The liquor at this temperature thus contained 5.24 lb alum (dry)/gallon.

4. Maximum doses at each depth interval, calculated earlier as lb alum/M^3 were converted to gallons/M^3 by dividing by 5.24 lb/gallon, as follows:

Depth Interval	lb Alum (dry)/M^3	gal/M^3
5-6 M	0.5076	0.09687
6-7 M	0.5568	0.10626
7-bottom	0.6041	0.11529

Therefore, the dose in gallons to any depth interval could be calculated by simply multiplying the volume of water in each interval by the appropriate unit volume dose.

ALUM APPLICATION

Based on earlier considerations (see Materials and Methods) it was decided to confine treatment to the hypolimnetic region of West Twin Lake. Although the depth of the metalimnetic -hypolimnetic boundary varies during the course of the summer period and could be affected by meteorologic factors, the center of the metalimnion was, on the average, located at a depth of 5 M. It is at this depth that the oxygen concentration generally drops below 1.0 PPM and total phosphorus concentration begins to increase. Therefore, it was decided to set the upper limit of the treatment zone at a depth of 5 M. This insured that those sediments exposed to oxygen concentrations of 1.0 PPM or less would receive aluminum hydroxide floc, and thus maximized the potential for controlling anaerobic sediment phosphorus release. The choice of this depth also allowed for maximal phosphorus removal during treatment.

West Twin was surveyed during the week prior to treatment to delineate the upper limit of the 5 M zone and to establish treatment quadrats of known area and volume. The perimeter of the upper limit of the 5 M zone was determined by sounding, using a graduated, weighted line and marked at the surface with white buoys. The area thus encircled represented the treatment area. The treatment area was bi-sected by a north-south transect located to divide the treatment zone into nearly equal parts (east zone = 159913 M^3 ; west zone = 171084 M^2). Two additional north-south transects were then established 100 M to either side of the first transect. All three transects were measured with surveyor's transit.

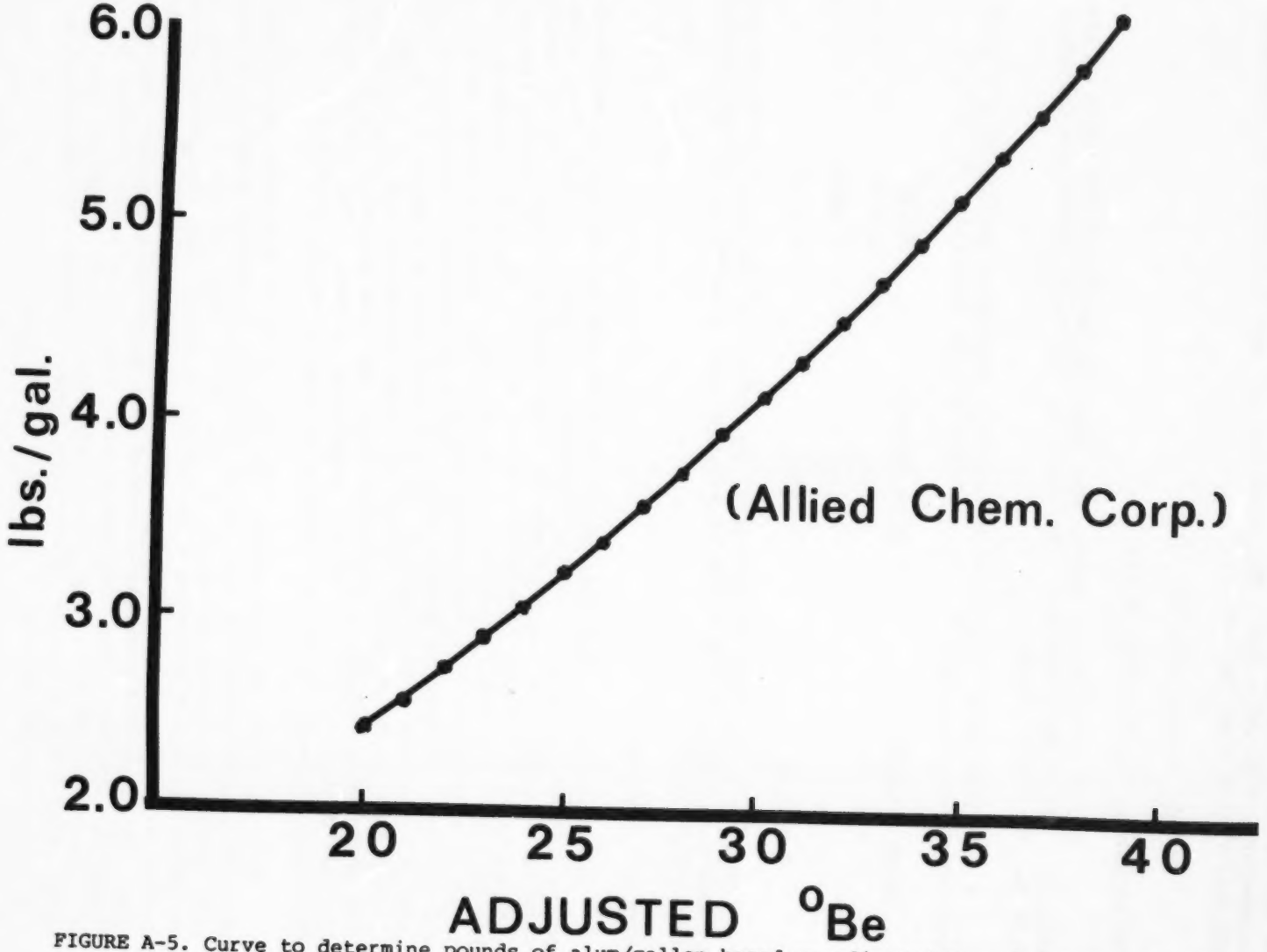

FIGURE A-5. Curve to determine pounds of alum/gallon based on adjusted Baume' (Anonymous, 1970)

Treatment quadrats were established by locating east-west transects at 50 M intervals along the original north-south transect. Coded buoys marked all intersections of east-west transects with either a north-south transect or the perimeter of the treatment zone. Specific quadrats, identifiable by a four number code, could be easily located by locating the appropriate four corner buoys. The correct location of each of the corner buoys was confirmed by triangulation from shore points.

Depth soundings were taken at each buoy. These readings were used to determine the volume of water below a depth of 5 M in each quadrat. Volumes of depth intervals uninterrupted by lake bottom were calculated by multiplying the area of the quadrat by one meter. Volumes of depth intervals in which the bottom occurred before the next even depth or passed through any part of the depth interval were calculated individually by calculating the volume of one or more geometric solids. Volumes below a depth of 7 meters were summed in each quadrat.

Gallonage required to treat each quadrat was calculated by multipling the volume of each depth interval by the appropriate unit dose and then summing the dose for all depth intervals. Quadrat doses are listed in Table A-6. The total dose (i.e.,the sum of all quadrat doses) applied to West Twin was 36,919 gallons. This total dose is higher than that calculated independently. That is, if the relationship between aluminum dose and alkalinity, the alkalinities of each depth interval, and the total volume of each depth interval (i.e., based on lake morphology and not quadrat morphology) are used instead, the required dose would be 34,444 gallons. Thus the calculations by quadrats resulted in an overdose of 2475 gallons, or 7.2%. This difference was probably due to the unusual geometric shape of the last partial meter of water in some of the quadrats. This probably led to some overestimations of volume of high alkalinity water.

EQUIPMENT FOR ALUM APPLICATION

The equipment for the treatment of West Twin Lake was modelled after that used in the Dollar Lake treatment (Kennedy, 1978). The system consisted of a storage facility, a delivery system, and application vehicles.
(1) Storage facility - Although the decision to use liquid alum simplified treatment procedures by not requiring equipment for mixing, it did require that adequate liquid storage be provided. An 18 foot diameter, 4 foot deep above-ground swimming pool with a maximum capacity of 7,600 gallons was used. The capacity of the pool was more than sufficient to hold one tank truck load of alum (approximately 4,000 gallons). The timing of truck arrivals was adjusted such that off-loading of alum occurred when the pool was approximately one third full, thus preventing a bottleneck at the pool site or an interruption in application.

The pool was constructed within 10 M of a paved road on a hill approximately 100 M north of the lake. A flat area was covered with a 8-10 cm layer of sand to level the site. The site was approximately 10-15 M above lake level, thus allowing for gravity-assisted pumping to the lake.

Table A-6. GALLONS OF ALUMINUM SULFATE APPLIED TO WEST TWIN LAKE

Quadrat	Gallonage	Quadrat	Gallonage
1	38	21	2986
2	234	22	2751
3	1436	23	1105
4	3077	24	791
5	1866	25	927
6	500	26	848
7	91	27	996
8	11	28	554
9	176	29	60
10	547	30	61
11	627	31	380
12	910	32	1220
13	2347	33	1070
14	2541	34	450
15	2282	35	274
16	855	36	430
17	595	37	843
18	208	38	786
19	181	39	211
20	1654		

(2) Delivery system - The delivery system consisted of three components: (1) a 12 hp, 2 1/2 inch pump, (2) a system of pipe leading from the pool-side pump out into the lake, and (3) a floating platform, which served as a loading point for application vehicles.

The pump was located at pool side with the intake port attached to a siphon laid over the rim of the pool and the output port attached to the delivery pipe. Although alum solutions are caustic there appeared to be no damage to the impeller, casing, or fittings after use. All of these parts were brass.

The piping system consisted of coupled 20 foot sections of 2 3/8" I.D. (1/4" wall) PVC rigid wall pipe floated under anchored 55 gallon drums spaced 10 M apart. Two and one half inch fire hose was used for the segment from the pump to the lake shore to avoid problems presented by uneven terrain. The pipe terminated at a butterfly valve located on an 8' by 8' platform situated in the lake and anchored on all four corners. A 20' section of flexible tubing (of the same diameter), connected to the outflow side of the valve, served for loading tanks on application vehicles.

(3) Application vehicles - Two barges were used for application of the alum. The barge design developed for the Dollar Lake treatment (Kennedy, 1978) was adopted with modifications. A series of pontoons, constructed by welding five capped 55 gal drums end-to-end, were laid side-by-side and lashed with 3/8" steel cable to a steel frame constructed from 4 inch channel iron. Five rows (25 drums) of pontoons were used for one barge, and six rows (30 drums) for the second. The long axes of the pontoons were oriented bow to stern. Two 250 gallon fuel storage tanks, the bottom drains of which were joined by 2 1/4" galvanized pipe, were strapped to the top of the steel frame, allowing a total payload of 500 gallons.

The buoyancy of one drum (subtracting its own weight) is approximately 400 lb. The total weight of the tanks (plus 500 gallons of alum at approximately 11.5 lb/gallon), structural steel, and miscellaneous on-board equipment was approximately 8,000 lbs. The buoyancy provided by five and six rows of pontoons was 10,000 and 12,000 lbs, respectively.

A seven meter long application manifold, constructed from PVC pipe (2 3/8" I.D., 1/4" wall), was suspended at a depth of 5 M below each barge by attaching it to a steel frame. The frame consisted of two A-frames, the legs of each being attached to the side of the barge at the bow and the stern. The apexes were joined below the barge by a cross-member equal in length to the width of the barge.

The manifold was attached to this cross-member, capped at the ends, and drilled with 1/4" holes, oriented bottom and back. A 1 1/2 hp gasoline pump, located on the deck of each barge, was fitted with a tee coupling at the intake, allowing alum from the tanks and lake water from just below the surface to be drawn simultaneously. The percent flow from each source, adjusted using two brass butterfly valves, was maintained at 50/50 during application to provide initial mixing before exiting the manifold. The only other mixing occurred as a result of turbulence created by the forward movement of the manifold. A clear plastic standpipe, graduated in gallons, allowed a constant monitoring of the quantity of alum in the tanks.

The barges were propelled by outboard motors (35 hp) fitted with work props, which provided adequate power. Steering and throttle controls were located just aft of the tanks to insure greatest visibility.

ALUM APPLICATION PROCEDURES

Treatment began on July 29 and required three full days to complete. A total of 12,693 gallons was added on day one, 12,555 gallons on day two, and 11,671 gallons on the third day. Operations started at approximately 0700 hr and continued till dark (2200 hr), giving an application rate (per barge) of between 0.26 hr/100 gallons and 0.24 hr/100 gallons. This rate includes time to load barge tanks and re-position equipment.

The platform which received alum from the shore and served as a "filling station" for the barges was positioned each day near the quadrats to be treated. Each platform location allowed the barges to approach from opposite sides of the platform. To be loaded, barges were moored, bow in. When the filling hose was in place and the line valve opened, instructions to begin pumping were radioed to the pump operator at pool side. Pumping was terminated when the tanks were nearly full and were allowed to completely fill by gravity flow. When it had been confirmed by radio that the pump was off, the valve was slowly closed.

Alternately colored floats spaced seven meters apart (the length of the manifolds) on a 50 meter line served as navigational aids. A line strung between the north and south buoy at the eastern end of the quadrat and another line, similarly marked with floats, was strung along the western end. The first float on each line was 3.5 meters from the end, with the remaining six buoys being spaced 7 meters apart. The barge moved back and forth across the quadrat by lining-up on like-colored floats, thus insuring even coverage of the area. Barge speeds were kept below 3 mph. When a quadrat was completed, the navigational lines and markers were moved by persons in smaller boats, while the barges returned for refilling of the tanks.

Surface water and alum, mixed 50/50, were pumped down to the manifold by an on-board 1 1/2 hp gasoline pump. At the end of each run, the pumps were allowed to pump water only while returning to the platform to flush out the steel pump casing.

Equipment dismantling and removal required approximately one week.

COST OF WEST TWIN ALUM APPLICATION

The cost of the West Twin treatment is itemized in Table A-7. Labor costs are not given, but the number of hours spent on construction and application are listed. The costs for equipment and alum are likely to increase over the years, but the labor hours could be greatly reduced by the employment of experienced personnel or by the development of a faster means of application.

Table A-7. COST FOR THE ALUMINUM SULFATE TREATMENT OF WEST TWIN LAKE

A.	Itemized Costs	
1.	Equipment (re-usable)	$ 4,058
2.	Supplies	646
3.	Rentals	262
4.	Aluminum sulfate	6,803
	TOTAL	$11,779
B.	Labor Hours	
1.	Total hours, construction and application	2,000
2.	Application (divided among 11 people over 3 days)	590